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BASELINE HUMAN HEALTH RISK ASSESSMENT FOR RECREATIONAL VISITORS AT RICHARDSON FLAT TAILINGS PARK CITY, SUMMIT COUNTY, UTAH

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EXECUTIVE SUMMARY

Site Description and Background

The Richardson Flats Tailing (RFT) Site is located 1.5 miles northeast of Park City, Utah occupying about 700 acres in a small valley in Summit County, Utah (Figure 1-1). The RFT site is part of the Park City Mining District where silver-laden ore was mined and milled from the Keetley Ontario Mine as well as other mining operations (RMC, 2001a). Tailings were deposited into an impoundment covering 160 acres of the 700 acre property just east of Silver Creek. Tailings were deposited to the impoundment from the mill by use of a slurry pipeline from 1975 through 1981. Mining and milling operations ended in 1982.

This document is a baseline human health risk assessment (BHHRA) for recreational users of the RFT site. The purpose of the document is to assess the health risks to visitors, from chemical contaminants in tailings and other environmental media present at this site. The results of this assessment are intended to help inform risk managers and the public about the level of health risk which is attributable to the contamination, to help determine the need for remedial action at the site, and to provide a basis for determining the levels of chemicals that can remain onsite and still be adequately protective of public health (USEPA 1989a).

Selection of Chemicals of Potential Concern

The Chemical of Potential Concern (COPC) were selected using a four step selection process as follows:

Step 1: Evaluation of Essential Nutrients
Step 2: Evaluation of Detection Frequencies
Step 3: Comparison with Background Concentrations
Step 4: Toxicity/Concentration Screen

Based on these steps, arsenic and lead were identified as COPCs and evaluated quantitatively in the site risk assessment.

Exposure Assessment

Land use at this site is limited to recreational purposes. In the future, it is expected the land use will remain recreational, and it is not envisioned that this property will be developed for residential purposes.

There are a wide variety of different recreational activities which people may engage in at this site, and hence there are a wide variety of different recreational exposure scenarios which might warrant evaluation. Two separate use scenarios were considered to serve as the representative populations evaluated:

- **low intensity users** such as, hikers, bikers, and picnickers
- **high intensity users** such as, horseback riders, ATV users, dirt-bikers, soccer and baseball players

The low intensity users were assumed to range in age from young children to adults, whereas the high intensity users were assumed to be an older (teenage to adult) population. Although there may be some instances where a child (1-6 years) may be a high intensity user, this scenario is not evaluated in the risk assessment. The risk assessment is based on the assumption that no further remedial or construction activities will occur at the site. That is, the activities listed will be assumed to occur on current contaminated site conditions, rather than on baseball and/or soccer fields created using clean fill material, sod and turf.

There are a number of pathways by which these recreational visitors may come into contact with contaminants in site media. The following exposure scenarios were judged to be of sufficient potential concern to warrant quantitative exposure and risk analysis at this site:

Population	Pathway
Low Intensity User	-Ingestion of Soil/Tailings -Ingestion of Surface Water -Dermal Exposure to Surface Water -Ingestion of Sediment -Inhalation of Particulates in Air (from wind erosion)
High Intensity User	-Ingestion of Soil/Tailings -Inhalation of Particulates in Air (from human disturbances and activity)

Quantification of Exposure and Risk from Arsenic

Methods

Risks to low- and high-intensity recreational visitors from exposure to arsenic in site media were evaluated according to standard USEPA methods.

All exposure and toxicity factors used for the varying exposure scenarios are presented in Chapter 5 of the risk assessment. The relative bioavailability of arsenic was assumed to be equal to the default value of 80% due to a lack of site-specific data.

Concentrations of Arsenic

Because the true mean concentration of a chemical within an Exposure Point cannot be calculated with certainty from a limited set of measurements, the USEPA recommends that the upper 95th confidence limit (UCL) of the arithmetic mean concentration be used as the Exposure Point Concentration (EPC) in calculating exposure and risk (USEPA 1992a). If the calculated UCL is higher than the highest measured value, then the maximum value is used as the EPC instead of the UCL (USEPA 1992a). In accord with this policy, EPCs were calculated for arsenic in each of the media types at this site. These values are summarized below:

Media	EPC for Arsenic
Sediment	200 mg/kg
Surface Water	0.012 mg/L
Soil/Tailings	55 mg/kg
Air- (High Intensity User)	0.000005 mg/m^3
Air- (Low Intensity User)	0.0000000016 mg/m^3

Noncancer and Cancer Risks

Noncancer risks are described in terms of the ratio of the dose at the site divided by a dose that is believed to be safe. This ratio is referred to as the Hazard Quotient (HQ). If the HQ is equal to or less than a value of 1, it is believed that there is no appreciable risk that noncancer health effects will occur. If an HQ exceeds 1, there is some possibility that noncancer effects may occur, although an HQ above 1 does not indicate an effect will definitely occur. However, the larger the HQ value, the more likely it is that an adverse health effect may occur.

Arsenic is listed by USEPA as an oral carcinogen. Risk of cancer from exposure to arsenic is described in terms of the probability that an exposed individual will develop cancer because of that exposure by age 70. The level of cancer risk that is of concern is a matter of individual, community and regulatory judgement. However, the USEPA typically considers risks below 1 in a million to be so small as to be negligible, and risks above 100 per million to be sufficiently large that some sort of action or intervention is usually needed.

Results

The following table presents both cancer and non-cancer risks for exposure to arsenic by both low- and high-intensity recreational users. As seen, for both low- and high-intensity users the total risks are below a Hazard Index of 1.0 for both average and RME exposure assumptions. The majority of the predicted risk is primarily attributable to ingestion of soils/tailings. Excess cancer risks were not found to exceed 100 cases per million for either low- or high-intensity recreational users under either average or RME exposure scenarios.

Receptor	Non-Cancer HI		Cancer Risk (per million)	
	Avg	RME	Avg	RME
Low Intensity User	0.01	0.09	<1	22
High Intensity User	0.006	0.05	<1	12

Uncertainties

Several assumptions used in the evaluation of risks from non-lead COPCs at this site may introduce uncertainty into the presented findings. Although in most cases, assumptions employed in the risk assessment process to deal with uncertainties are intentionally conservative; that is, they are more likely to lead to an overestimate rather than an underestimate of risk, it is nevertheless important for risk managers and the public to take these uncertainties into account when interpreting the risk conclusions derived for this site.

Uncertainties presented in the risk assessment include: uncertainty in concentration estimates. uncertainty in human intakes, uncertainty in toxicity values, uncertainty in absorption from soil, uncertainty from pathways not evaluated and uncertainty in summing risks across exposure pathways.

Quantification of Exposure and Risk from Lead

Methods

Risks from lead are usually evaluated by estimation of the blood lead levels in exposed individuals and comparison of those blood lead values to an appropriate health-based guideline. In the case of lead exposure, the population of chief concern is young children (age 0-84 months), due to the type of health effects that occur in this age bracket. The USEPA and CDC have set as a goal that there should be no more than a 5% chance that a child should have a blood lead value over 10 ug/dL. For convenience, the probability of exceeding a blood lead value of 10 ug/dL is referred to as P10.

Blood lead levels in an exposed population of children may either be measured directly, or may be calculated using a mathematical model. Because no measured blood data were available, the modeling approach was utilized at this site. Both young children (less than 7 years of age) and adults were evaluated for exposure to lead in the low intensity recreational scenario. The modeling approaches used to evaluated these two distinct age groups are explained below. Under the high intensity scenario only exposure to teenagers and adults was evaluated.

Risks to Young Children

The USEPA has developed an integrated exposure, uptake and biokinetic (IEUBK) model to assess the risks of lead exposure in residential children (0 to 6 years). This model requires as input point estimates of the average concentration of lead in various environmental media in residential properties at the site, and the average amount of these media contacted by a child living at the site. These data are used to estimate the average blood lead value in an exposed child. Then, a distribution of blood lead values is estimated by assuming a lognormal distribution and applying an estimated geometric standard deviation (GSD).

For this site, two simulations were run using the IEUBK model. The first evaluated risks to a hypothetical nearby resident. The second simulation was used to address the risk observed when the hypothetical residential child engaged in low-intensity recreational activities at the site. By comparing the two simulations and resulting predictions of blood lead concentrations, the excess risk attributable to the recreational exposure can be identified, in order to judge whether the risks to any random child participating in site-based recreational activities are within health based goals.

The resulting predictions of the IEUBK model for these two scenarios are shown below. As seen, children who engage in low intensity recreational activities at this site have higher predicted blood lead levels than those with no recreational exposure. However, the geometric mean values are relatively low and children engaging in recreational activities have under a 5% chance of exceeding a blood lead value of 10 ug/dL using a GSD value of either 1.4 or 1.6.

Scenario	GSD = 1.4		GSD = 1.6	
	Geometric Mean Blood Lead (ug/dL)	P10	Geometric Mean Blood Lead (ug/dL)	P10
Residential Only	1.8	<0.01%	1.8	0.01%
Residential + Recreational	2.0	<0.01%	2.0	0.01%

These results indicate that current risks to recreational child visitors from lead is likely to be well below USEPA's health-based goal at this site.

Risks to Older Children and Adults

The risks to teenage and adult recreational visitors (low and high intensity) from exposure to lead in site media were evaluated using the Bowers model. This model predicts the blood lead level in an adult exposed to lead by summing the "baseline" blood lead level (PbB_0) (that which would occur in the absence of any above-average site-related exposures) with the increment in blood lead that is expected as a result of increased exposure due to contact with a lead-contaminated site medium. This model was run in accord with guidance developed by USEPA's Technical Workgroup for Lead (USEPA, 1996b).

For low intensity visitors, the geometric mean blood lead concentration was predicted to be 1.4 ug/dL with a PbB_{95} value of 4.8 ug/dL. For high intensity visitors, the geometric mean blood lead concentration was predicted to be 1.5 ug/dL with a PbB_{95} value of 5.1 ug/dL. The USEPA has not yet issued formal guidance on the blood lead level that is considered appropriate for protecting the health of pregnant women or other adults. Therefore, these results can be interpreted using a health criterion that there should be no more than a 5% chance that the blood level of a fetus will be above 10 ug/dL. This is equivalent to a blood lead concentration of 11.1 ug/dL in the pregnant adult. A comparison of the 95[th] percentile blood lead levels predicted for site recreational visitors shows that recreational use at this site is not predicted to result in blood lead levels which exceed a target concentration of 11.1 ug/dL under either low- or high-intensity use scenarios.

Uncertainties

Several assumptions used in the evaluation of lead risks at this site may introduce uncertainty into the presented findings. Although in most cases, assumptions employed in the risk assessment process to deal with uncertainties are intentionally conservative; that is, they are more likely to lead to an overestimate rather than an underestimate of risk, it is nevertheless important for risk managers and the public to take these uncertainties into account when interpreting the risk conclusions derived for this site. Uncertainties presented in the risk assessment include: uncertainty in lead concentrations estimates, uncertainty in lead absorption from soil, and uncertainty in the modeling approach.

Conclusions

The results of risk calculations for arsenic presented in this report indicate that for all evaluated scenarios (low-intensity, high-intensity, CTE, RME) non-cancer risks are below a Hazard Index of one. Additionally, all cancer risks were estimated to be within or below USEPA's acceptable risk range of one in a million to one in 100,000.

Risks from lead exposure were evaluated at this site using both the IEUBK model (children) and the Bowers model (teenagers and adults). Both models resulted in predictions of blood lead levels that were below a 5% probability of exceeding a blood lead level of 10 ug/dL.

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LIST OF ACRONYMS AND ABBREVIATIONS

ACGIH	American Conference of Governmental Industrial Hygienists
AF	Absorption Fraction
ASARCO	American Smelting and Refining Company
AT	Averaging Time
ATSDR	Agency for Toxic Substances and Disease Registry
ATV	All-terrain Vehicle
BHHRA	Baseline Human Health Risk Assessment
BKSF	Biokinetic Slope Factor
BW	Body Weight
C	Concentration
CDC	Centers for Disease Control
CEPA	California Environmental Protection Agency
COPC	Contaminant of Potential Concern
CTE	Central Tendency Exposure
DI	Daily Intake
E&E	Ecology & Environment, Inc.
ED	Exposure Duration
EF	Exposure Frequency
EPC	Exposure Point Concentration
FS	Fraction Contributed from Site
GSD	Geometric Statndard Deviation
HI	Hazard Index
HIF	Human Intake Factor
HQ	Hazard Quotient
HRS	Hazard Ranking System
IEUBK	Integrated Exposure Uptake and Biokinetic
IR	Ingestion Rate
IRIS	Integrated Risk Information System
LOAEL	Lowest Observed Adverse Effect Level
NOAEL	No Observed Adverse Effect Level
NPL	National Priorities List
PbB	Blood Lead Concentration
PbS	Soil Lead Concentration
PC	Permiability Constant
PCV	Park City Ventures
PEF	Particulate Emissions Factor
PPM	Parts Per Million
RBA	Relative Bioavailablity
RBC	Risk-based Concentration
RCRA	Resource Conservation and Recovery Act
RfD	Reference Dose
RFT	Richardson Flat Tailings
RI/FS	Remedial Investigation/Feasibility Study

LIST OF ACRONYMS AND ABBREVIATIONS (Continued)

RMC	Resource Management Consultants
RME	Reasonable Maximum Exposure
SA	Surface Area
SF	Slope Factor
STORET	EPA's Storage and Retrieval System
TAL	Target Analyte List
TDS	Total Dissolved Solids
TMDL	Total Maximum Daily Load
TSS	Total Suspended Solids
TWA	Time-weighted Average
UCL	Upper Confidence Limit
UPCM	United Park City Mines
USEPA	United States Environmental Protection Agency
XRF	X-Ray Fluorescence

1.0 INTRODUCTION

1.1 Site Description

The Richardson Flats Tailing (RFT) Site is located 1.5 miles northeast of Park City, Utah occupying about 700 acres in a small valley in Summit County, Utah (Figure 1-1). The RFT site is part of the Park City Mining District where silver-laden ore was mined and milled from the Keetley Ontario Mine as well as other mining operations (RMC, 2001a). Tailings were deposited into an impoundment covering 160 acres of the 700 acre property just east of Silver Creek. Tailings were deposited to the impoundment from the mill by use of a slurry pipeline from 1975 through 1981. Mining and milling operations ended in 1982. A detailed description of the site history is presented in Section 2.

1.2 Purpose and Scope

This document is a baseline human health risk assessment (BHHRA) for recreational users of the RFT site. The purpose of the document is to assess the health risks to visitors, from chemical contaminants in tailings and other environmental media present at this site. The results of this assessment are intended to help inform risk managers and the public about the level of health risk which is attributable to the contamination, to help determine the need for remedial action at the site, and to provide a basis for determining the levels of chemicals that can remain onsite and still be adequately protective of public health (USEPA 1989a).

The methods used to evaluate risks to humans and the environment employed in this assessment are consistent with current guidelines provided by the USEPA for use at Superfund sites (USEPA 1989a, 1991b, 1993a).

1.2 Organization

In addition to this introduction, this report is organized into the following sections:

Section 2	This section provides the site characterization, which includes the site location, description, regulatory history, and environmental setting.
Section 3	This section provides a summary of the available data on the levels of chemical contaminants (metals) in site media, and identifies which of these chemicals are of potential health concern to area residents.
Section 4	This section discusses how visitors may be exposed to site-related chemicals, now or in the future, and identifies exposure scenarios that are considered to be of potential concern.

Section 5	This section assesses the level of exposure and risk to humans from non-lead chemicals of potential concern at this site. This includes 1) a description of methods used to quantify exposure to these chemicals, 2) data on the toxicity of these chemicals to humans, 3) calculation of the level of noncancer and cancer risk that may occur as a result of exposure to these chemicals in site soils, and 4) a discussion of the uncertainties which limit confidence in the assessment.
Section 6	This section assesses the level of exposure and risk to area visitors from lead in site soils. This includes 1) a description of the toxic effects of lead, 2) a summary of the method used by USEPA to evaluate risks from lead, 3) a summary of the estimated risks at this site attributable to lead in site soils, and 4) a discussion of the uncertainties which limit confidence in the assessment.
Section 7	This section summarizes the overall findings presented in Sections 5 and 6.
Section 8	This section provides full citations for USEPA guidance documents, site-specific studies, and scientific publications referenced in the risk assessment.

2.0 SITE CHARACTERIZATION

This section contains the location, description, regulatory history and environmental setting of the RFT Site. This information originated in the RFT Screening Ecological Risk Assessment (USEPA, 2002a), but has been reiterated in this document for individuals who may not be familiar with the site background.

2.1 Site Location

As discussed in Section 1, the RFT Site is a 700 acre property located in a small valley in Summit County, approximately 1.5 miles northeast of Park City, Utah (Figure 1-1). This site is part of the larger Park City Mining District where silver-laden ore was mined and milled from the Keetley Ontario Mine as well as other mining operations (RMC, 2001a). Tailings from these operations were deposited onsite into an impoundment covering approximately 160 acres of RFT property. These tailings were deposited to the impoundment just east of Silver Creek mill by use of a slurry pipeline from 1975 through 1981. Mining and milling operations ended in 1982.

2.2 Site Description

Tailings were first placed on the RFT Site prior to 1950 (RMC, 2000a). Historical aerial photos confirm that tailings have been present at the flood plain tailings pile as early as 1953 (USEPA, 1991a). The mill tailings present consist mostly of sand-sized particles of carbonate rock with some minerals containing silver, lead, zinc and other metals. Few specific details are available concerning the configuration and operation of the historic tailings pond (prior to 1950) but certain elements are apparent. From time to time, tailings were transported to the Site through three distinct low areas on the southeast portion of the Site. Over the course of time, tailings materials settled out into the low areas that were ultimately left outside and south of the present impoundment area constructed in 1973 to 1974 (RMC, 2001b).

In 1970, Park City Ventures (PCV), a joint venture partnership between Anaconda Copper Company and American Smelting and Refining Company (ASARCO) entered into a lease agreement with United Park to use the Site for the disposal of additional mill tailings generated from renewed mining in the area. PCV contracted with Dames & Moore to provide construction specifications for reconstruction of the Site for continued use as a tailings impoundment (Dames & Moore, 1974). The state of Utah approved the Dames & Moore plan and the current impoundment area was constructed in 1974 (RMC, 2000a). Before disposing of tailings on the Site, PCV installed a large earthen embankment along the western edge of the existing tailings impoundment and constructed perimeter containment dike structures along the southern and eastern borders of the impoundment to allow storage of additional tailings. PCV also installed a diversion ditch system along the higher slopes north of the impoundment and outside of the containment dike along the east and south perimeter of the impoundment to prevent surface runoff from surrounding land from entering the impoundment (RMC, 2001b). Dames & Moore recommended that specially engineered seepage control devices be installed at the base of the main embankment. PCV did not follow this recommendation (Dames & Moore, 1974).

PCV conveyed tailings to the impoundment by a slurry pipeline from its mill facility located south of the Site. Over the course of operation, approximately 420,000 tons of tailings were disposed of at the Site. PCV failed to follow recommendations for disposal of the slurry in the impoundment (to place tailings along the perimeter of the impoundment and move towards the center) and placed a large volume of tailings near the center of the impoundment in a large, high-profile, cone-shaped feature. After cessation of operations in 1982, the presence of the cone-shaped feature resulted in prevailing winds form cutting into the tailings and the tailings becoming wind-borne (RMC, 2001b).

The RFT Site is currently under the ownership of United Park City Mines (UPCM) (RMC, 2000a). UPCM is a consolidation of Silver King Coalition Mines Company and Park Utah Consolidated Mines Company, formed in 1953 (RMC, 2000a).

2.2.1 Sources

There are two known sources of contamination at the RFT Site. These include the tailings impoundment previously described and a flood plain tailings pile. The flood plains tailings pile is located immediately west of the tailings impoundment and covers about 6 acres along the banks of Silver Creek (USEPA, 1991a). This source is reported to be located on the western side of Silver Creek about 300 feet upstream of the confluence of Silver Creek with the wetland area and extends from there for about 2,500 feet upstream. The USEPA and the State of Utah have both observed tailings entering Silver Creek from the flood plain tailings pile (USEPA, 1991a). According to analyses performed in 1985 and 1989, the flood plain tailings pile contains arsenic, cadmium, copper, lead, mercury, silver, and zinc (USEPA, 1991a).

2.2.2 Site Features

The Focused Remedial Investigation/Feasibility Study (RI/FS) Workplan prepared by RMC in May 2000, provides detailed information on the RFT Site features. Information pertaining to the main embankment and containment dikes, the diversion ditches and off-impoundment tailings is summarized in the following subsections.

Main Embankment and Containment Dikes

The majority of the tailings at the RFT Site are contained in a closed basin, with a large, earth, embankment in place along the western edge of the Site. The "main embankment" is vegetated and is approximately 40 feet wide at the top, 800 feet long, and has a maximum height of 25 feet. This embankment is designed to allow water to seep from the impoundment to relieve hydraulic pressure on the embankment. Currently, surface water is present in the form of a seep located near the north end of the base. A series of man-made containment dikes contain the tailings along the southern and eastern perimeter of the impoundment. The northern edge of the impoundment is naturally higher than the perimeter dikes (RMC, 2000a).

Diversion Ditches

A diversion ditch system borders the north, south, and east sides of the impoundment to prevent runoff from the surrounding land from entering the impoundment. Precipitation falling on the impoundment area creates a limited volume of seasonal surface water. The north diversion ditch collects snowmelt and storm water runoff from upslope, undisturbed areas north of the impoundment and carries it in an easterly direction towards origin of the south diversion ditch. An unnamed ephemeral drainage to the southeast of the impoundment also enters the south diversion ditch at this point. Additional water from spring snowmelt and storm water runoff enters the south diversion ditch from other areas lying south of the impoundment at a point near the southeast corner of the diversion ditch structure. Water in the south diversion ditch flows from east to west and ultimately empties into Silver Creek just upstream of Highway 189 near the north border of the Site. Water flow from the south diversion ditch into Silver Creek occurs during the higher water periods of the year (RMC, 2000a).

Off-Impoundment Tailings

Additional tailings materials are present outside and to the south of the current impoundment area. During historic operations of the tailings pond, tailings accumulated in three naturally low areas adjacent to the property that eventually became the impoundment. In the 1970s, when PCV constructed the perimeter dike and diversion ditch along the south perimeter of the impoundment, tailings present in the three low areas were left in place, outside of the present impoundment. Starting in 1983, United Park reportedly covered most of these tailings outside of the current impoundment with a low permeability, vegetated soil cover. Other types of clean fill material, imported from construction work in Park City, were also used to cover the tailings outside of the impoundment. The cover in some of these areas is reported to be as thick as 10 to 15 feet (RMC, 2000a). However, recent surveys of off-impoundment cover soils indicate that at some locations soil cover is absent leaving exposed surface tailings and in other places the soil cover is less than a few inches (RMC, 2001a).

2.2.3 Site Activities

UPCM and others have conducted certain efforts at the RFT Site to support investigation of integrity or closure. These activities are briefly described in the following subsections.

Impoundment Integrity Analyses

Noranda Mining, Inc. (Noranda) leased the RFT Property from UPCM in 1980 (RMC, 2000a). Shortly after Noranda entered into the lease agreement, Dames & Moore was contracted to conduct an impoundment integrity investigation. Although several construction flaws were noted, including the oversteeping of the main embankment along various locations, Dames & Moore concluded that the main embankment and containment dikes were in no immediate threat of failure. Dames & Moore once again recommended the installation of seepage control systems at the base of the main embankment (RMC, 2000a). Noranda did not follow this recommendation. Noranda disposed of 70,000 tons of additional tailings material and ceased operations in 1982. No new tailings have been placed at the Site since that time (RMC, 2000a).

Soil Cover of Tailings

Starting in 1983, UPCM began placing soil cover on tailings outside of the impoundment, located in three low areas south of the south diversion ditch. By 1985, the tailings impoundment had dried out enough in certain areas to support heavy equipment and UPCM began installing soil cover material over those portions. The cover soils are reported to be clay-rich and came from both the Park City area and from within the RFT Site (RMC, 2000a).

Between 1985 and 1988, UPCM also placed soil cover around the cone shaped tailings structure inside the impoundment area at locations where it had dried out enough to support heavy equipment. The primary objective of placing the soil cover was to prevent prevailing winds from cutting into the cone-shaped tailings

By 1988, this work was completed and UPCM began a more aggressive program to cover all exposed tailings. It is reported that at least 12 inches of low-permeability, clay cover material was placed in the impoundment and that the soil cover was then vegetated (RMC, 2000a). More recent inspection of the cover soils at the main impoundment and off-impoundment indicate a shallow soil cover in some areas (less than 12 inches) and no soil cover in other locations (RMC, 2001a).

By 1992, repairs to soil cover work were completed (RMC, 2000a). Shortly after completion, E&E (1993) completed a soil depth survey within the impoundment and an inspection of the main embankment. X-Ray Fluorescence (XRF) was used to confirm the visual contrast between top soil and the tailings below (E&E, 1993). E&E (1993) determined that on average, cover soils varied between less than 6 inches and 14 inches in depth. Areas in which cover soils were known to be more than 3 feet in depth were not surveyed. For the 29 locations studied, one exhibited exposed tailings. As a result, UPCM placed additional soil in this area (RMC, 2000a). More recent soil cover surveys for the main impoundment, however, indicate that at some locations the soil cover is less than 12 inches in depth (RMC, 2001a; 2001b).

Wedge Buttress Reinforcement

In an effort to correct the over-steepened portions of the main embankment, UPCM proposes to design the installation of a wedge buttress. The buttress will enhance the long-term effectiveness of the final closure remedy for the Site. UPCM will evaluate the condition of the main embankment during the RI/FS, and then prepare construction design specifications as part of the final remedial design process. Data from the seep located at the base of the main embankment may need to be gathered in order to develop an appropriate wedge buttress design (RMC, 2000a).

Fencing

In the mid 1980's, UPCM installed a fence along most of the Site boundary, including the entire impoundment and much of the property south of the impoundment. The fence was placed to restrict access to the Site. UPCM reports it will maintain the fence in good repair and will continue to control site access until such time limited access is no longer necessary (RMC, 2000a).

Diversion Ditch Reconstruction

In 1992 and 1993, UPCM reconstructed the south diversion ditch by decreasing the slope of its banks from nearly vertical to a more gradual slope. UPCM placed a clay soil cover over the re-sloped banks down to and including areas of the banks underwater. The existing ditch banks were re-vegetated and the bottom of the ditch was not disturbed during these efforts. In May of 1999, United Park reconstructed the north diversion ditch along its entire length in the same manner (RMC, 2000a).

2.3 Regulatory History

The RFT Site was first proposed for the National Priorities List (NPL) on June 24, 1988. The original Hazard Ranking System (HRS) score of 50.23 was based on surface water and air migration pathways (USEPA, 1991a). Areas evaluated in the HRS included the impoundment and adjacent areas (USEPA, 1991a). Based on public comments, the site was dropped from consideration for the NPL on February 11, 1991 (USEPA, 1991a). The HRS scoring criteria for surface water migration pathways were revised in 1992. The USEPA is currently proposing the site for a second NPL consideration under the revised HRS (USEPA, 1991a). Along with the impoundment area and adjacent areas, the new proposal includes the Park City Municipal Landfill and the Silver Creek flood plain area (RMC, 2000a).

2.4 Site Environmental Setting

2.4.1 Topography and Surrounding Land Use

The site is located in a rural area whose topography is characterized by a broad valley with undeveloped rangeland. Silver Creek is located within a few hundred feet from the main tailings impoundment. This perennial stream drains other historic tailing ponds in the Park City area (Mason, 1989). Silver Creek originates in an upper mountain zone where access is limited to recreational users. As Silver Creek passes through Park City and in to the surrounding suburban areas, the land use is primarily residential and commercial, changing to recreational and agricultural downstream to its confluence with the Weber River (RMC, 2001a).

2.4.2 Geology and Hydrogeology

Geology

The RFT Site is located in the Wasatch Range Section of the Middle Rocky Mountain Physiographic Province in north-central Utah in an area composed of a complex fold and thrust belt that is covered over with igneous rock (RMC, 2000a; 2000b). The sedimentary bedrock, which dates to the Paleozoic and Mesozoic age, is covered by a thick layer of extruded igneous rock that dips approximately 25 to 60 degrees to the north and strikes northeast-southwest (Bromfield and Crittenden, 1971). Tertiary gravels and igneous rocks cover the Mesozoic sedimentary rocks (RMC, 2001a). There are no known faults near the RFT Site.

Alluvial and colluvial sediments lie 30 to 50 feet deep beneath the tailings on site. These sediments are a product of the erosion of neighboring and underlying igneous extrusions. Borehole data have shown that these sediments consist of: 2-5 feet of soft, organic, and clay rich topsoil; 1-30 feet of mixed fine-grained silt and clay; 4 feet of sand and gravel; highly weather, volcanic breccia which is composed of soft, tight, sandy and silty clay grading to harder fractured volcanic rock (RMC, 2000b). The unconsolidated valley fill is reported to range in thickness from a few feet adjacent to hills and mountains to at least 260 feet, centrally in valleys (Mason, 1989)

Hydrogeology

In 1999, UPCM contracted Weston Engineering, Inc. (Weston) to conduct a hydogeological survey of the site. The hydrogeology in the area consists of shallow alluvial aquifers located in the alluvial and colluvial material as well as the deeper Silver Creek Breccia bedrock aquifer located in the Keetley volcanics (RMC, 2000b). The shallow aquifers are found fifteen to thirty feet below the ground surface in gravelly clay. The shallow aquifers' hydraulic gradients parallel topography (south to north) except at the southern boundary of the tailings embankment where flow changes to the northwest due to diversion ditches. The hydrogeology of the Site area has been described in a separate report (Weston, 1999).

Hydrology

Silver Creek flows approximately 500 feet from the main embankment along the west edge of the Site (RMC, 2000a). The headwaters of Silver Creek are comprised of three major drainages in the Upper Silver Creek Watershed; the Ontario Canyon, the Empire Canyon and Deer Valley. Flows from Ontario and Empire Canyons occur in the late spring to early summer months in response to snowmelt and rainfall, while Deer Valley flows appear to be perennial and originate from snowmelt and springs (RMC, 2000b). Surface water runoffs for this watershed are lower than those of comparable mountain watersheds which are less fractured and may have a more developed layer of unconsolidated materials (Brooks et al., 1998). Overall, runoff and precipitation flows from Empire and Ontario Canyons are low compared to the substantially large flow contributed by Deer Valley (USEPA, 2001a). The major influence on water flow in Silver Creek near the RFT Site is the Pace-Homer (Dority Springs) Ditch, which derives most of its flow from groundwater (USEPA, 2001a). The outflow from the Pace-Homer Ditch enters Silver Creek at several locations across the Prospector Square area. Significant riparian zones and wetlands exist near the RFT Site in areas that historically consisted of accumulated tailings piles.

2.4.3 Climate

Richardson Flat is located in north-central Utah. The average monthly precipitation is approximately 3.64 inches with an average annual precipitation of 43.68 inches (The Weather Channel, 2001). The average monthly temperature ranges from a low of 13.9°F (December) to a high of 81.5°F (July) (Western Regional Climate Center, 2002). Elevations near the RFT Site range from 6,930 to 9,075 feet above sea level (RMC, 2000b).

3.0 DATA SUMMARY AND EVALUATION

The BHHRA is based on the available analytical and physical data from investigations completed within the RFT Site area. A summary of the raw data is provided as Appendix A. These results represent the known nature and extent of contamination and are used as the basis of the BHHRA. The BHHRA is based only on analytical data from within or adjacent to the site. The study area boundary is shown in Figure 3-1.

3.1 Tailings Data

As previously discussed, contamination at the RFT Site originated from the deposition of tailings within and outside of an impoundment. In July 1989, one tailings sample from the main impoundment area (stratified depths from 1-18 inches) and five tailings samples (0-6 inches) from flood plain areas were collected and data were presented in the Hazard Ranking System (USEPA, 1991a). These samples were analyzed for total arsenic, cadmium, copper, lead, mercury, silver and zinc.

In May 2001, RMC collected tailings samples from the three locations within the impoundment at 1 foot depth intervals (beginning from the bottom of the cover soils to a depth of 5 feet). Samples were analyzed for aluminum, antimony, arsenic, cadmium, chromium, copper, iron, lead, mercury, selenium, silver, and zinc. These samples were collected to evaluate the long-term fate of metals in tailings and the chemical stability of the tailings (RMC, 2001a).

Tailings disposal is also present in areas located outside the impoundment, but the spatial extent of these areas are not well defined. In June 2001, RMC collected tailings samples from locations south of the south diversion ditch in an effort to determine the extent of tailings disposal. This study was also completed to evaluate soil cover thickness, and if the tailings were contributing to zinc concentrations in the south diversion ditch. Samples were analyzed for aluminum, antimony, arsenic, cadmium, chromium, copper, iron, lead, mercury, selenium, silver, and zinc.

3.2 Soils Data

3.2.1 On-Impoundment Soils

In August 1992, Ecology & Environment, Inc. (E&E), under direction from USEPA, investigated the RFT Site with respect to immediate threats to human health or the environment. The depth of soil cover was determined at 29 locations on the impoundment (based on an approximate grid pattern of 400 ft by 400 ft). At six of these locations, samples were analyzed for Target Analyte List (TAL) metals. Each of the samples, with the exception of sample RF-SO-3, are representative of cover soils on the impoundment in 1992. Sample RF-SO-3, was collected in an area of salt grass not yet covered by UPCM and is representative of tailings (E&E, 1993). Subsequently, UPCM placed additional soil cover in areas with thin cover (as identified by E&E, 1993) and on other areas to support site closure efforts (RMC, 2001a).

Currently, the tailings impoundment is reported to be covered with soil and vegetation with no areas of exposed tailings (RMC, 2001a). However, the extent, thickness, and chemical characteristics of the cover soils are not well defined. In May 2001, RMC collected 41 cover soils from 6 transects based on a 500 ft by 500 ft grid across the impoundment at a depth of 0-2 inches (distinct locations are identified as A through I).

Additional depth samples, ranging from 5 to 18 inches, were collected at 11 of these locations. All samples were analyzed for arsenic and lead with 20% of the samples analyzed for all RCRA metals.

3.2.2 Background Soils

In order to determine the concentrations of metals in areas not affected by wind-blown tailings from the RFT Site, RMC collected background samples from areas not impacted by tailings deposition. It is important to note that these samples are representative of anthropogenic, non-site related levels, and do not represent "pristine" (not influenced by human activity) environmental levels. Therefore, these samples were not utilized in the BHHRA.

3.3 Surface Water Data

Surface water data were compiled from five sources including E&E (1993), Utah water quality monitoring, USEPA (2001a), UPCM surface water monitoring, and RMC monthly sampling. A description of the surface water data from each source is provided in the following subsections.

For the purposes of conducting the BHHRA, surface water data from Silver Creek were limited to those stations adjacent to the RFT site boundaries. Upstream/downstream locations were excluded from further evaluation. Water data for the south diversion ditch are limited to samples collected after ditch reconstruction (1993 to present).

<u>Ecology & Environment, Inc. (1993)</u>

In August 1992, E&E collected surface water samples from Silver Creek and the south diversion ditch. Six samples were collected along Silver Creek (RF-SW-1 to RF-SW-6) and two samples were collected from the south diversion ditch (RF-SW-7 and RF-SW-8). On-site and adjacent samples included in this assessment were RF-SW-3, 4, 5, 7 and 8. Water data for the south diversion ditch (RF-SW-7 and RF-SW-8) are limited to samples collected after ditch reconstruction (1993 to present).

<u>Utah Water Quality Monitoring *(STORET)*</u>

Water quality monitoring data for several stations along Silver Creek were obtained electronically from an USEPA STORET download query (Modernized Version). Data is available from nine locations on Silver Creek of which one is located adjacent to the RFT site. Samples are collected and analyzed monthly for water quality parameters such as total hardness, pH, and temperature, as well as total recoverable and dissolved metals including arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc. Information for the Silver Creek station located adjacent to the RFT site is provided in the following text table.

Station ID	Location Description	Latitude	Longitude	Sampling Dates
492685	Silver Creek at US40 Crossing east of Park City	40.683000	-111.456000	02-May-75 to17-Jun-99

USEPA (2001a) Silver Creek Watershed Sampling

In 2000, USEPA completed an investigation of the Silver Creek watershed to better characterize the sources of heavy metals and to evaluate the total maximum daily load (TMDL). A total of 31 surface water sampling locations are available from the watershed study for Silver Creek and its headwaters in Empire Canyon, Ontario Canyon, Deer Valley. For the purposes of the BHHRA only data from locations on or adjacent to the site are used for the risk evaluation. Surface water samples for USC-3 and USC-4 were collected from the south diversion ditch on the RFT Site. Samples were collected in May and September 2000, respectively, to account for high (peak spring runoff) and low flow (fall or winter seasons).
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UPCM Monitoring

Since 1975, UPCM has collected surface water samples from the south diversion ditch (N5), and Silver Creek upstream (N4) and downstream (N6) of the confluence with the south diversion ditch. Surface water samples were collected monthly (usually from April to November) and analyzed for copper, cyanide, lead, mercury, manganese, zinc, total suspended solids (TSS) and total dissolved solids (TDS). Surface water data collected prior to April 1982 were not available. Surface water data for the south diversion ditch (N5) are limited to samples collected after ditch reconstruction (1993 to present). All data from this report were used except for three samples, where results for Hg were excluded because the reported values appeared anomalous compared to all others (RMC, 2002). These three excluded values are listed below:

Station	Date	Mercury (mg/L)
Upstream Silver Creek (N4)	7/8/84	0.9
	9/6/84	2.0
Downstream Silver Creek (N6)	9/6/84	2.1

RMC Monthly Sampling (RMC, 2001c)

Since May 1999, RMC has collected monthly surface water from several locations along Silver Creek, the south diversion ditch, the unnamed drainages flowing into the south diversion ditch, and ponded areas at the RFT Site. Specific locations are identified in and detailed station information is summarized in the following text table. Surface water samples were analyzed for total recoverable and dissolved TAL metals and water quality parameters.

Station ID	Location Description	Sampling Dates
RF-2	South diversion ditch	19-May-99 to 7-May-01
RF-3	Unnamed drainage flowing into the south diversion ditch	19-May-99 only
RF-3-2	Unnamed drainage flowing into the south diversion ditch	4-Apr-01 to 5-Jun-01
RF-4	South diversion ditch	19-May-99 to 9-Jul-01
RF-5	South diversion ditch	19-May-99 to 7-Aug-01
RF-6	South diversion ditch	19-May-99 to 18-Sep-00
RF-6-2	South diversion ditch	9-Jun-99 to 3-Dec-01
RF-7	Silver Creek upstream of confluence with south diversion ditch	19-May-99 to 7-Nov-00
RF-7-2	Silver Creek upstream of confluence with south diversion ditch	9-Jun-99 to 3-Dec-01
RF-8	Silver Creek downstream of the confluence with south diversion ditch	19-May-99 to 3-Dec-01
RF-9	Ponded water on the tailings impoundment	19-May-99 only
RF-10	Unnamed drainage flowing into south diversion ditch	9-Jun-99 only

3.4 Sediment Data

Sediment data are compiled for the BHHRA from three separate sources including E&E (1993), USEPA (2001a) and RMC monthly sampling.

Use of surface water data for the south diversion ditch in the BHHRA is limited to samples collected after ditch bank modification (1993 to present). This limitation is not, however, placed on the use of sediment data. During reconstruction, UPCM did not disturb the bottom of the ditch bed (RMC, 2001a) thus the existing sediments were not disturbed and constraining use of the data set is not necessary.

As with the surface water data set, only Silver Creek sediments collected adjacent to the site were utilized in the risk assessment.

Ecology & Environment, Inc. (1993)

In August 1992, E&E collected four sediment samples (RF-SD-01 to RF-SD-04) from the south diversion ditch "wetlands" area located at the base of the main embankment and Silver Creek. Water flow through this wetlands area is primarily from the south diversion ditch, although some seepage from the impoundment area may influence the flow and chemistry (E&E, 1993). Based on the ratios of chemicals in tailings compared to those in the wetlands sediments, E&E concluded that the sediments in the wetlands area are tailings material from the impoundment (E&E, 1993).

USEPA (2001a) Watershed Sampling

USEPA collected sediment samples from 16 locations in the Silver Creek watershed. These samples were staggered across the watershed and co-located with specific surface water sampling sites to determine the relative level of metals throughout the system and evaluate interactions with surface water (USEPA, 2001a). At each location, both a surface and sub-surface (0-12 inches) sample was collected and analyzed for heavy metals. Because the BHHRA was limited to on-site and adjacent sampling locations, none of these analyses were included in this assessment.

RMC Monthly Sampling (RMC, 2001c)

In May 2001, RMC sampled sediments at six locations (RF-SD-1 to RF-SD-6) along the length of the south diversion ditch at a depth of 0 to 6 inches. These samples were collected to evaluate the long-term effectiveness of the wetland system to remove metals in the water and to aid in the determination of the source of metals in water flowing from the diversion ditch (RMC, 2001a).

3.5 Seep Data

Because the main embankment is designed to allow water to seep from the impoundment to relieve hydraulic pressure, it is likely that metals leach from tailings into groundwater at the RFT Site. At the RFT Site, a small seep (flow of gallons per day) is located at the northern base of the main embankment (RMC, 2000a). Currently, no water or sediment data exist for this seep.

3.6 Groundwater Data

Since 1973, PCV and UPCM have collecting groundwater data quarterly from monitoring wells MW-1, MW-2, and MW-3 (RMC, 2000a). After their installation in 1976, PCV also began collecting groundwater from wells MW-4, MW-5, MW-6. E&E began collecting additional groundwater data in 1984 from a well (RT-1) installed up gradient of the main embankment. E&E also sampled the two existing down gradient monitoring wells MW-1 and either MW-5 or MW-6 . [It is unclear as to which well, MW-5 or MW-6, was sampled.] Well MW-2 was buried during the installation of wells MW-4, MW-5, MW-6 in 1976. The USEPA contracted E&E in 1992 to collect ground water samples from three additional locations (RF-GW-04, RF-

GW-05, and RF-GW-09). Consumption of groundwater is not a complete pathway for the recreational visitors at this site, therefore these data were not utilized in this assessment.

3.7 Air Data

In July 1986, air monitoring at RFT documented detectable concentrations of arsenic, cadmium, lead, and zinc in air. Since that time, cover soil was placed over the tailings area. Subsequent air monitoring was conducted during June10-11, 1992, at five locations around the perimeter of the site. Arsenic, cadmium and lead were not detected (detection limits not specified) in any of the samples. Zinc was detected at low concentrations (0.1 ug/m^3) at four of the five monitoring stations (E&E 1993). Because of the lack of quantitative values, unknown detection limits, these data are not considered suitable for the risk assessment. Additionally, the short duration of the sampling period may or may not be representative of the spatial and temporal variability of ambient air concentrations at the site.

3.8 Biological Tissue Data

At the time of the BHHRA, the analyses of contaminant concentrations in biological tissues (aquatic or terrestrial) were not available from existing data reports and literature.

3.9 Summary of Analytical Data

Table 3-1 provides a summary of the analytical data available for the BHHRA. This table compares the analytical parameters available for the environmental media sampled and analyzed. As previously described, there are eight sources of sampling data including: RMC (2000a), USEPA (1991a); E&E (1993); USEPA (2001a); RMC(2001a); RMC (2001c); UPCM and STORET. These programs do not have one common list of analytes for all environmental media. Table 3-1 provides a side-by-side comparison of the parameters available for each media type from each source of sampling data. Summary statistics for the data used in this assessment are provided in Table 3-2.

3.10 Selection of COPCs

Step 1. Evaluation of Essential Nutrients

In accord with USEPA guidance (1989a, 1994a), chemicals that are normal constituents of the body and the diet and are required for good health may be eliminated unless there is evidence that site-specific releases have elevated concentrations in a range where intakes would be potentially toxic. Of the chemicals analyzed in soils and water at this site, 14 are classified as essential nutrients (calcium, cobalt, chloride, chromium, copper, fluoride, iron, magnesium, manganese, phosphorus, potassium, selenium, sodium, and zinc). Therefore, the assumed recreational intakes of these 14 constituents in site media were compared to their corresponding toxicity value or safe nutritive level as provided in USEPA (1994a). The parameters used to calculate the recreational intake values are presented in Appendix B. These values were then multiplied by the maximum detected concentration of a chemical in each media to obtain a daily intake for that chemical. This intake was then divided by the screening value provided by USEPA (1994a) to determine if the chemical could be eliminated from further analysis based on an observed ratio of less than 1.0 (i.e., predicted intake does not exceed safe level).

Results are summarized in Table 3-3. As shown, all of the beneficial chemicals analyzed in sediments and surface water can be eliminated from further evaluation. For soil and tailings, only four beneficial chemicals were analyzed. All four (Chromium III, Copper, Selenium, Zinc) are below safe levels and can also be eliminated as potential COPCs.

Step 2: Evaluation of Detection Frequencies

A contaminant with a detection frequency of ≥5% is carried through the toxicity/concentration screening process (Step 3). Chemicals having detection frequencies of <5% are usually assumed to be non-site related and are generally not evaluated as COPCs. However, it is important to ensure that the detection limit for such chemicals would have been adequate to detect the chemical if it were present at levels of human health concern. In sediments all chemicals analyzed were detected at frequencies greater than 5% and all of the detection limits were deemed adequate. Of the chemicals analyzed in surface water, three were observed with a detection frequency below 5%: silver, thallium, vanadium. Table 3-4 shows that the detection limits for these chemicals were adequate for risk assessment purposes. Thus, silver, thallium, and vanadium were eliminated as COPCs in surface water. In sediment, soil and tailings, no chemicals were observed to have a detection frequency of less than 5%. Therefore, All of the chemicals will be carried through for further evaluation as COPCs.

Step 3: Comparison with Background Concentrations

Concentrations of analyzed metals in site soils and tailings were compared to their published background ranges (Dragun, 1988; Shacklette and Boerngen, 1984; ATSDR, 1997). This comparison is presented in Table 3-5. As shown, both the average and maximum concentration of barium fall squarely within the ranges reported for the United States. Therefore, it was eliminated from further analysis as a COPC at this site. The other chemicals (arsenic, cadmium, lead, mercury and silver) were either clearly higher or not obviously within the reported background levels, and were carried further through the COPC selection process.

Step 4: Toxicity/Concentration Screen

The final step used to evaluate COPCs at this site was a toxicity/concentration screen conducted in accord with USEPA (1994a) guidance. This step involves comparing the maximum reported concentration of a chemical in a medium to an appropriate Risk-Based Concentration (RBC). RBCs are media-specific health-based levels which if exceeded, could indicate that there is a potential for adverse health effects to occur as a result of exposure. If the maximum concentration value is less than the RBC, the chemical does not pose an unacceptable health risk and can be eliminated as a COPC. [Note: This is true providing that the chemical does not exceed any relevant ARAR values.]

The RBCs used in this evaluation were calculated using intake parameters associated with recreational visitors (see Appendix B for intake parameters). Further details of the RBC calculations are presented in Appendix C. RBC's were calculated for water, sediment, and soil/tailings. The value of each RBC depends on the specified Target Risk level. In accord with the goal that the COPC selection process should be conservative, the Target Risk levels used in this evaluation are 1E-06 for carcinogenic chemicals and a hazard quotient (HQ) of 0.1 for noncarcinogenic chemicals.

Table 3-6 lists the maximum concentration and RBC values used to evaluate each chemical in sediment, surface water, and soil/tailings and identifies those chemicals which were not eliminated from further consideration at this step.

Summary

The COPC screening process identified arsenic and lead for further quantitative evaluation in the risk assessment at this site.

4.0 EXPOSURE ASSESSMENT

Exposure is the process by which humans come into contact with chemicals in the environment. In general, humans can be exposed to chemicals in a variety of environmental media (e.g., soil, dust, water, air, food), and these exposures can occur through one or more of several pathways (ingestion, dermal contact, inhalation). Section 4.2 provides a discussion of possible pathways by which recreational users might come into contact with contaminants present in site media. Sections 5 and 6 describe the basic methods used to estimate the amount of chemical exposure (non-lead and lead) which humans may receive from direct and indirect contact with contaminants derived from outdoor soil.

4.1 Conceptual Site Model

Figure 4-1 presents a generalized conceptual site model showing the main pathways by which contaminants from current or former mining activities and other sources might come into contact with people exposed within the RFT site boundary. Exposure scenarios that are considered most likely to be of concern are shown in Figure 4-1 by a solid circle, while pathways which are judged to contribute only minor exposures are shown by a cross-hatched circle. Incomplete pathways (i.e., those which are not thought to occur) are shown by open circles.

4.1.1 Potential Sources

As discussed in Section 2, there are two known sources of contamination at the RFT Site. These include the primary onsite tailings impoundment and a flood plain tailings pile.

4.1.2 Migration Pathways

The current medium of chief concern is soil and tailings materials. Metals in these materials tend to have relatively low mobility and are most likely to move by wind-blown transport of suspended particles in air, surface run-off from nearby piles, or by hauling of bulk material from one location to another.

4.1.3 Exposed Populations and Potential Exposure Scenarios

Land use at this site is currently limited to recreational purposes. In the future, it is expected the land use will remain recreational, and it is not envisioned that this property will be developed for residential purposes.

There are a wide variety of different recreational activities which people may engage in at this site, and hence there are a wide variety of different recreational exposure scenarios which might warrant evaluation. Two separate use scenarios were considered to serve as the representative populations evaluated:

- low intensity users such as, hikers, bikers, and picnickers
- high intensity users such as, horseback riders, ATV users, dirt-bikers, soccer and baseball players

The risk assessment is based on the assumption that no further remedial or construction activities will occur at the site. That is, the activities listed will be assumed to occur on current contaminated site conditions, rather than on baseball and/or soccer fields created using clean fill material, sod and turf.

4.2 Pathway Screening

4.2.1 Recreational Exposure - Low Intensity Users

Several pathways of exposure were reviewed for the low intensity recreational user. The low intensity user is an individual who visits the site for the purposes of activities such as hiking, biking, picnicking. It is thought that on occasion these visitors may also engage in activities at surface water locations, such as wading and splashing. The exposure pathways identified for these low intensity users are discussed in more detail below.

Incidental Ingestion of Soil/Tailings

Few people intentionally ingest soil. However, it is believed that most people (especially children) do ingest small amounts of soil that adhere to the hands or other objects placed in the mouth. This exposure pathway is often one of the most important routes of human intake, so it was selected for quantitative evaluation.

Dermal Contact with Soil

Visitors can get contaminated soil on their skin while engaging in recreational activities at the site. Even though information is limited on the rate and extent of dermal absorption of metals in soil across the skin, most scientists consider that this pathway is likely to be minor in comparison to the amount of exposure that occurs by soil and dust ingestion. This view is based on the following concepts: 1) most people do not have extensive and frequent direct contact with soil, 2) most metals tend to bind to soils, reducing the likelihood that they would dissociate from the soil and cross the skin, and 3) ionic species such as metals have a relatively low tendency to cross the skin even when contact does occur. Screening calculations (presented in Appendix D) support the conclusion that dermal absorption of metals from dermal contact with soil is likely to be relatively minor compared to the oral pathway, and omission of this pathway is not likely to lead to a substantial underestimate of exposure or risk. Based on these considerations, along with a lack of data to allow reliable estimation of dermal uptake of metals from soil, Region 8 generally recommends that dermal exposure to metals in soils not be evaluated quantitatively (USEPA 1995). Therefore, this pathway was not evaluated quantitatively in this risk assessment.

Inhalation of Soil/Tailings in Air

Low-intensity users may be exposed to particles of contaminated soil or dust that become re-suspended in air from wind erosion or by human disturbances and activity. Visitors may breathe those particles while engaged in activities at the site. The low intensity user is not likely to be involved in activities that result in intensive contact with site soils that would result in re-suspension of contaminated material in air from human disturbances. Therefore, this pathway was not evaluated quantitatively. However, the low intensity user may be exposed to particulates re-suspended in air from wind erosion while visiting the site. Therefore, this pathway was selected for further quantitative evaluation.

Ingestion of Site Biota

Silver Creek is a potential location for fishing, and anglers who catch fish from reaches with significant water and/or sediment contamination may be exposed via ingestion of the fish. Similarly, hunters who harvest game animals (deer, waterfowl, etc.) from locations with significant contaminant levels in soil, vegetation or water may be exposed via ingestion of the game. Although it is considered plausible that this pathway might contribute a considerable fraction of the total exposure, especially for individuals who rely on local fish or game as a main component of their diet, no data are available on contaminant levels in these media. Therefore, this pathway was not evaluated. Although data were not available to evaluate this pathway, total exposure is not likely to be significantly underestimated, as the chemicals of concern for the site (arsenic and lead) do not accumulate in fish tissues consumed by humans.

Ingestion of Surface Water

In warm weather, Silver Creek is a potential location for recreational activities such as wading and splashing. Although it is not expected that recreational visitors intentionally drink water from the river, these activities can lead to incidental ingestion of water, so this pathway was selected for quantitative evaluation.

Dermal Contact with Surface Water

Recreational visitors to the site may wade in the water at Silver Creek or in onsite wetlands areas, so dermal contact with surface water is likely (at least during warm weather). Therefore, the dermal exposure pathway for recreational visitors was evaluated quantitatively.

Contact with Sediments

People who enter the river or recreate in the onsite wetlands or drainage ditch areas may come into contact with sediments in the river bed, and exposure could presumably occur either by incidental ingestion and/or by dermal contact. However, because contact with sediments is associated with being in a water source, any material that gets on the hands or skin is likely to be largely washed off by the water. Therefore, dermal exposure to sediments was not evaluated quantitatively, however, incidental ingestion of these sediments was retained as a quantitative pathway of concern.

4.2.2 Recreational Exposure - High Intensity Users

Several pathways of exposure were reviewed for the high intensity recreational user. The high intensity user is an individual who visits the site for the purposes of activities such as horseback riding, dirt-bike and ATV riding, and playing soccer and/or baseball. It is thought that this group of recreational visitors is likely to have more intensive contact with site soils than the low intensity users. Additionally, this visitor is not expected to recreate in site surface waters. The exposure pathways identified for these high intensity users are discussed in more detail below.

Incidental Ingestion of Soil/Tailings

Few people intentionally ingest soil. However, it is believed that most people (especially children) do ingest small amounts of soil that adhere to the hands or other objects placed in the mouth. This exposure pathway is often one of the most important routes of human intake, so it was selected for quantitative evaluation.

Inhalation of Soil/Tailings in Air

Particles of contaminated soil or dust may become re-suspended in air from wind erosion or by human disturbances and activity. Visitors may breathe those particles while engaged in activities at the site. Because high intensity activities may result in higher concentrations of contaminants being re-suspended in air, this pathway was selected for further quantitative evaluation. Although a high intensity user may also be exposed to particles re-suspended in air from wind erosion while visiting the site, the concentrations of contaminants in air from wind erosion are likely to be small relative to the concentrations re-suspended from human disturbances. Therefore, this pathway was not selected for further quantitative evaluation.

4.3 Summary of Pathways of Principal Concern

Based on the evaluations above, the following exposure scenarios are judged to be of sufficient potential concern to warrant quantitative exposure and risk analysis:

Population	Pathway
Low Intensity User	-Ingestion of Soil/Tailings -Ingestion of Surface Water -Dermal Exposure to Surface Water -Ingestion of Sediment -Inhalation of Particulates in Air (from wind erosion)
High Intensity User	-Ingestion of Soil/Tailings -Inhalation of Particulates in Air (from human disturbances and activity)

5.0 QUANTIFICATION OF EXPOSURE AND RISK FROM ARSENIC

5.1 Quantification of Exposure

5.1.1 Basic Equation

The magnitude of human exposure to chemicals in an environmental medium is described in terms of the average daily intake (DI), which is the amount of chemical which comes into contact with the body by ingestion, inhalation, or dermal contact. The general equation for calculating the daily intake from contact with an environmental medium is (USEPA 1989a):

$$DI = C \times IR \times EF \times ED \times RBA / (BW \times AT)$$

where:

DI	=	daily intake of chemical (mg/kg-d)
C	=	concentration of chemical in an environmental medium (e.g., mg/kg)
IR	=	intake rate of the environmental medium (e.g., kg/day)
EF	=	exposure frequency (days/yr)
ED	=	exposure duration (years)
RBA	=	relative bioavailability of chemical in site medium
BW	=	body weight (kg)
AT	=	averaging time (days)

For mathematical and computational convenience, this equation is often written as:

$$DI = C \times HIF \times RBA$$

where:

HIF = "Human Intake Factor". For soil and dust ingestion, the units of HIF are kg/kg-day. The value of HIF is given by:

$$HIF = IR \times EF \times ED / (BW \times AT)$$

There is often wide variability in the amount of contact between different individuals within a population. Thus, human contact with an environmental media is best thought of as a distribution of possible values rather than a specific value. Usually, emphasis is placed on two different portions of this distribution:

- Average or Central Tendency Exposure (CTE) refers to individuals who have average or typical intake of environmental media.
- Upper Bound or Reasonable Maximum Exposure (RME) refers to people who are at the high end of the exposure distribution (approximately the 95th percentile). The RME scenario is intended to assess exposures that are higher than average, but are still within a realistic range of exposure.

The following sections list the exposure equations and exposure parameters used in the BHHRA for evaluation of low and high intensity recreational visitors by inhalation of particulates, incidental ingestion of soil/tailings, ingestion of and dermal contact with surface water (low intensity only), or incidental ingestion of sediment (low intensity only), along with the resulting HIF terms for CTE and RME exposure.

central tendency exposure

5.1.2 Exposure Equations and Parameters for the Low Intensity Recreational Visitor

Both children (1-6 years) and adult recreational visitors have potential exposure pathways of soil/tailing ingestion and inhalation of particulates during low intensity activities and may be expected on a more infrequent basis to engage in recreational activities where exposure to sediments and surface water are plausible. Health endpoints include both cancer (via chronic exposure) and non-cancer health effects.

Soil/Tailings Ingestion

Both chronic and lifetime average intake rates are time-weighted to account for the possibility that an adult may begin exposure as a child (USEPA 1989a, 1991b, 1993a), as follows:

$$TWA - DIs = Cs\left(\frac{IRc}{BWc} \bullet \frac{EFc \bullet EDc}{(ATc + ATa)} + \frac{IRa}{BWa} \bullet \frac{EFa \bullet EDa}{(ATc + ATa)}\right)$$

where:

$TWA\text{-}DI_s$ = Time-weighted Daily Intake from ingestion of soil/tailings (mg/kg-d)
C_s = Concentration of chemical in soil/tailings (mg/kg)
IR = Intake rate (kg/day) when a child (IR_c) or an adult (IR_a)
BW = Body weight (kg) when a child (BW_c) or an adult (BW_a)
EF = Exposure frequency (days/yr) when a child (EF_c) or an adult (EF_a)
ED = Exposure duration (years) when a child (ED_c) or an adult (ED_a)
AT = Averaging time (days) while a child (AT_c) or an adult (AT_a)

Default values and assumptions recommended by USEPA (1989a, 1991b, 1993a) for evaluation of exposure to soil/tailings are listed below. There are no data on ingestion rates of tailings by children or adults while engaged in recreational activities at this site. Therefore, based on professional judgment, ingestion rates of soil/tailings of 50 mg/event and 100 mg/event are assumed for adult and child RME low intensity visitors respectively. For CTE visitors, these values were assumed to be half of that attributable to the RME exposure (25 mg/day and 50 mg/day). Due to the lack of site specific data on the frequency of recreational use of the Richardson Flat Tailings Site, an open space usage survey in Jefferson County, Colorado (Jefferson County Open Space Department, 1996) were used to estimate the exposure frequency (EF) for recreational visitors at the Richardson Flats Tailings Site. During 1996, 779 individuals were interviewed and asked to quantify the number of times per year they visited Open Space Parks in Jefferson County. The arithmetic mean (39 visits/year) and 90th percentile (100 visits/year) of the total number of visits per year were calculated from the survey results and are used as the CTE and RME exposure frequency assumptions, respectively, for the Richardson Flats Site. The CTE and RME exposure frequencies were multiplied by an additional parameter, fraction of exposure at the site (FS), to adjust for the potential use of additional open spaces, other than the Richardson Flats Site, for recreation. In the absence of any site-specific data, the CTE and RME values for the FS parameter were set to 0.5 and 1.0, respectively, based on professional judgement. Theses values are thought to be appropriate for both CTE and RME scenarios by assuming that 50% and 100% of all recreational visits, respectively, occur at the Richardson Flats Tailings Site. Thus, 19.5 visits/year (CTE) and 100 visits per year (RME) are used as the exposure frequency assumptions at the site.

Exposure Parameters for Soil/Tailings Ingestion	CTE		RME	
	Child	Adult	Child	Adult
IR (kg/event)	50	25	100	50
BW (kg)	15	70	15	70
EF (events/year)	19.5	19.5	100	100
ED (years)	2	7	6	24
AT (non-cancer effects) (days)	2*365	7*365	6*365	24*365
AT (cancer effects) (days)	--	70*365	--	70*365

Based on the exposure parameters above, the HIFs for exposure of children and adults to soil/tailings are as follows:

Recreational Exposure to Soil/Tailings	HIF (kg/kg-d)	
	CTE	RME
TWA-chronic (non-cancer)	5.4E-08	5.2E-07
TWA-lifetime (cancer)	7.0E-09	2.2E-07

<u>Inhalation of Particulates</u>

The basic equation recommended by EPA (1989a) for evaluating exposure from inhalation of a chemical in air is:

$$TWA-DIa = Ca\left(\frac{IRc}{BWc}\bullet\frac{ETc\bullet EFc\bullet EDc}{(ATc+ATa)}+\frac{IRa}{BWa}\bullet\frac{ETa\bullet EFa\bullet EDa}{(ATc+ATa)}\right)$$

where:

$TWA\text{-}DI_a$ = Time-weighted Daily Intake from inhalation of a chemical in air (mg/kg-d)
C_a = Concentration of chemical in air (mg/m^3)
IR = Breathing rate of air (m^3/hour) when a child (IR_c) or an adult (IR_a)

ET = Exposure time (hours/day) when a child (ET_c) or an adult (ET_a)
EF = Exposure frequency (days/year) when a child (EF_c) or an adult (EF_a)
ED = Exposure duration (years) when a child (ED_c) or an adult (ED_a)
AT = Averaging time (days) while a child (AT_c) or an adult (AT_a)
BW = Body weight (kg) when a child (BW_c) or an adult (BW_a)
AT = Averaging time (days)

Default values and assumptions recommended by USEPA (1989a, 1991b, 1993a) for evaluation of exposure to particulates in air are listed below. Inhalation rates of 1.6 m^3/hr for children and 2.4 m^3/hr for adults are based on the average of medium and heavy activity inhalation rates for these age groups. This information is from the 1997 Exposure Factors Handbook and was used as inputs in the Rocky Flats Task 3 Report (USEPA, 2001b). The Exposure Time was based on the 1995 Boulder County open space survey (Boulder County Open Space Operations, 1995) of time spent on site (19% < 1 hour, 71% 1-3 hours, 9% 4-6 hours, and 1% >7 hours). Values of 1.5 and 2.5 hours/day were selected for the CTE and RME exposures, respectively. Although this information pertains to a different site, the values are judged to be applicable at Richardson Flats. The exposure frequency is estimated to be 19.5 days per year for CTE individuals and 100 days per year for RME individuals, based on the mean (39 visits per year for CTE) and 90th percentile (100 visits per year for RME) of visits to Jefferson County Open Space (Jefferson County Open Space Department, 1996) and the assumption that 50% (CTE) and 100% (RME) of all visits occur at the Richardson Flats site.

Exposure Parameters for Inhalation of Particulates	CTE		RME	
	Child	Adult	Child	Adult
IR (m^3/hr)	1.6	2.4	1.6	2.4
BW (kg)	15	70	15	70
ET (hr/day)	1.5	1.5	2.5	2.5
EF (days/yr)	19.5	19.5	100	100
ED (years)	2	7	6	24
AT (non-cancer effects) (days)	2*365	7*365	6*365	24*365
AT (cancer effects) (days)	--	70*365	--	70*365

Based on the exposure parameters above, the HIFs for exposure of children and adults to particulates are as follows:

Recreational Exposure to Particulates	HIF ($m^3/kg\text{-}d$)	
	CTE	RME
TWA-chronic (non-cancer)	4.0E-03	3.3E-02
TWA-lifetime (cancer)	5.2E-04	1.4E-02

Ingestion of Sediments

The basic equation used evaluating exposure from incidental ingestion of sediments by recreational visitors while visiting water areas is as follows. Both chronic and lifetime average intake rates are time-weighted to account for the possibility that an adult may begin exposure as a child (USEPA 1989a, 1991b, 1993a):

$$TWA - DIs = Cs\left(\frac{IRc}{BWc} \bullet \frac{EFc \bullet EDc}{(ATc + ATa)} + \frac{IRa}{BWa} \bullet \frac{EFa \bullet EDa}{(ATc + ATa)}\right)$$

where:

$TWA\text{-}DI_s$ = Time-weighted Daily Intake from ingestion of sediment (mg/kg-d)
C_s = Concentration of chemical in sediment (mg/kg)
IR = Intake rate (kg/day) when a child (IR_c) or an adult (IR_a)
BW = Body weight (kg) when a child (BW_c) or an adult (BW_a)
EF = Exposure frequency (days/yr) when a child (EF_c) or an adult (EF_a)
ED = Exposure duration (years) when a child (ED_c) or an adult (ED_a)
AT = Averaging time (days) while a child (AT_c) or an adult (AT_a)

Default values and assumptions recommended by USEPA (1989a, 1991b, 1993a) for evaluation of exposure by the ingestion of sediments are listed below. There are no data on ingestion rates of sediments by visitors while engaged in recreational activities along the river or in ponded water areas at the site. Therefore, in the absence of data, ingestion rates of soil/tailings of 25 mg/day and 50 mg/day are assumed for adult and child RME visitors respectively. For CTE visitors, these values were assumed to be half of that attributable to the RME exposure (12.5 mg/day and 25 mg/day). This is equivalent to half of the quantity consumed by the low intensity recreational visitor from soil/tailings ingestion. The exposure frequency is estimated to be 2 days per year for CTE individuals and 10 days per year for RME individuals, based on the assumption that the low intensity visitor is exposed to these media during 1 out of every 10 standard visits (4 visits per year (CTE) and 10 visits per year (RME)) and that 50% (CTE) and 100% (RME) of all visits occur at the Richardson Flats site.

The exposure parameters are summarized below:

Exposure Parameters for Ingestion of Sediments	CTE		RME	
	Child	Adult	Child	Adult
IR (kg/day)	25	12.5	50	25
BW (kg)	15	70	15	70
EF (days/year)	2	2	10	10
ED (years)	2	7	6	24
AT (non-cancer effects) (days)	2*365	7*365	6*365	24*365
AT (cancer effects) (days)	--	70*365	--	70*365

Based on these exposure parameters, the HIF values for exposure of visitors to sediments are as follows:

Recreational Exposure to Sediments	HIF (kg/kg-d)	
	Average	RME
Chronic (non-cancer)	2.8E-09	2.6E-08
Lifetime (cancer)	3.6E-10	1.1E-08

Dermal Contact with Surface Water

The basic equation recommended by USEPA (1989a) for evaluation of dermal exposure to a chemical dissolved in water is as follows. Both chronic and lifetime average intake rates are time-weighted to account for the possibility that an adult may begin exposure as a child (USEPA 1989a, 1991b, 1993a):

$$ADsw = Csw\left(\frac{SAc \bullet PC \bullet ETc \bullet 1E-03}{BWc} \bullet \frac{EFc \bullet EDc}{(ATc + ATa)} + \frac{SAa \bullet PC \bullet ETa \bullet 1E-03}{BWa} \bullet \frac{EFa \bullet EDa}{(ATc + ATa)}\right)$$

where:

AD_{sw} = Absorbed dose from dermal contact with surface water (mg/kg-d)
C_{sw} = Concentration of chemical in surface water (mg/L)
SA = Surface area exposed (cm^2) for child (SA_c) or adult (SA_a)
PC = Chemical-specific permeability constant (cm/hr)
ET = Exposure time (hr/day) for child (ET_c) or adult (ET_a)

1E-03 = Conversion factor (L/cm^3)
EF = Exposure frequency (days/yr) child (EF_c) or adult (EF_a)
ED = Exposure duration (yrs) for child (ED_c) or adult (ED_a)
BW = Body weight (kg) child (BW_c) or adult (BW_a)
AT = Averaging time (days) for child (AT_c) or adult (AT_a)

Default values and assumptions recommended by USEPA (1989a, 1991b, 1993a) for evaluation of exposure by dermal contact with surface water are listed below. It is assumed that dermal exposure of a recreation visitor to water occurs mainly while wading near the river edge or ponded areas. and that dermal contact is mainly restricted to the lower extremities (upper and lower legs and feet) as well as the hands. The surface area for these body parts in children and adults is the 50th percentile for hands, arms, and lower legs (USEPA, 1997) (SAF, 2000). No site-specific data on recreation exposure frequency or duration of wading activities per trip are available, so values of 2 (CTE) to 10 (RME) days/year, and 0.5 (CTE) to 1.5 (RME) hours/day are assumed. The exposure time is based on the FE Warren site (SAF, 2000), where estimated time spent in surface waters were evaluated. The exposure frequency is based on the assumption that the low intensity visitor is exposed to these media during 1 out of every 10 standard visits (4 visits per year (CTE) and 10 visits per year (RME)) and that 50% (CTE) and 100% (RME) of all visits occur at the Richardson Flats site. The value of PC is chemical specific, and few measured values are available for metals. Therefore, the USEPA (1992b) suggests using a PC value of 1E-03 cm/hr as a conservative estimate.

Exposure Parameters for Dermal Contact with Surface Water	CTE		RME	
	Child	Adult	Child	Adult
SA (cm^2)	3,800	5,000	3,800	5,000
PC (cm/hr)	1E-03	1E-03	1E-03	1E-03
BW (kg)	15	70	15	70
ET (hours/day)	0.5	0.5	1.5	1.5
EF (days/year)	2	2	10	10
ED (years)	2	7	6	24
AT (non-cancer effects) (days)	2*365	7*365	6*365	24*365
AT (cancer effects) (days)	–	70*365	--	70*365

Based on these exposure parameters, the HIF values for dermal exposure of low intensity recreational visitors to surface water are as follows:

Recreational Exposure for Dermal Contact with Surface Water	HIF (kg/kg-d)	
	Average	RME
Chronic (non-cancer)	3.1E-07	4.4E-06
Lifetime (cancer)	3.9E-08	1.9E-06

Ingestion of Surface Water

The basic equation for evaluation of exposure from ingestion of surface water while participating in water-based recreational activities is as follows. Both chronic and lifetime average intake rates are time-weighted to account for the possibility that an adult may begin exposure as a child (USEPA 1989a, 1991b, 1993a):

$$TWA - DIw = Cw\left(\frac{IRc}{BWc} \bullet \frac{ETc \bullet EFc \bullet EDc}{(ATc + ATa)} + \frac{IRa}{BWa} \bullet \frac{ETa \bullet EFa \bullet EDa}{(ATc + ATa)}\right)$$

where:

$TWA\text{-}DI_s$ = Time-weighted Daily Intake from ingestion of water (mg/kg-d)
C_s = Concentration of chemical in surface water (mg/L)
IR = Intake rate (L/day) when a child (IR_c) or an adult (IR_a)
BW = Body weight (kg) when a child (BW_c) or an adult (BW_a)
ET = Exposure time (hours/day) when a child (ET_c) or an adult (ET_a)
EF = Exposure frequency (days/yr) when a child (EF_c) or an adult (EF_a)
ED = Exposure duration (years) when a child (ED_c) or an adult (ED_a)
AT = Averaging time (days) while a child (AT_c) or an adult (AT_a)

Default values and assumptions recommended by USEPA (1989a, 1991b, 1993a) for evaluation of exposure by dermal contact with surface water are listed below. The RME intake rate for incidental water ingestion by recreational visitors of 30 mL/hour (RME) is the basis for the 10 mL/day value proposed in the Draft Water Quality Criteria Methodology Revisions (USEPA, 1998). Splashing or hand-to face contact while wading might result in only a very small amount of water in or near the mouth. For the CTE exposure scenario, the USEPA (1989a) default of 50 mL/hour for incidental ingestion during swimming is thought to be too high under this scenario. Based on this reasoning, a CTE value of 5 mL/hour (10% of the recommended default) was assumed. The exposure frequency is estimated to be 2 days per year for CTE individuals and 10 days per year for RME individuals, based on the assumption that the low intensity visitor is exposed to these media during 1 out of every 10 standard visits (4 visits per year (CTE) and 10 visits per year (RME)) and that 50% (CTE) and 100% (RME) of all visits occur at the Richardson Flats site.

Exposure Parameters for Ingestion of Surface Water	CTE		RME	
	Child	Adult	Child	Adult
IR (mL/hour)	5	5	30	30
BW (kg)	15	70	15	70
ET (hours/day)	0.5	0.5	1.5	1.5
EF (days/year)	2	2	10	10
ED (years)	2	7	6	24
AT (non-cancer effects) (days)	2*365	7*365	6*365	24*365
AT (cancer effects) (days)	--	70*365	--	70*365

Based on these exposure parameters, the HIF values for ingestion of surface water by recreational visitors are as follows:

Recreational Exposure for Ingestion of Surface Water	HIF (L/kg-d)	
	CTE	RME
Chronic (non-cancer)	3.6E-07	2.2E-05
Lifetime (cancer)	4.6E-08	9.6E-06

5.1.3 Exposure Equations and Parameters for the High Intensity Recreational Visitor

Adult recreational visitors have potential exposure pathways of soil/tailing ingestion and inhalation of particulates during high intensity activities (e.g. horseback riding, ATV use, dirt-biking, soccer and baseball). Health endpoints include both cancer (via chronic exposure) and non-cancer health effects.

Soil/Tailings Ingestion

The basic equation used for evaluating exposure from incidental ingestion of tailings or contaminated soil by recreational visitors is as follows:

$$DIs = Cs\left(\frac{IR}{BW}\right)\left(\frac{EF \bullet ED}{AT}\right)$$

where:

DI_s	=	Daily intake of chemical from ingestion of soil/tailings (mg/kg-d)
C_s	=	Concentration of chemical in soil/tailings (mg/kg)
IR_s	=	Intake rate (kg/event)
BW	=	Body weight (kg)
EF	=	Exposure frequency (days/year)
ED	=	Exposure duration (years)
AT	=	Averaging time (days)

Default values and assumptions recommended by USEPA (1989a, 1991b, 1993a) for evaluation of exposure by incidental ingestion of soil/tailings are listed below. There are no data on ingestion rates of soil or tailings by adults while engaged in high intensity recreational activities at this site. Therefore, based on professional judgment, ingestion rates of soil/tailings of 50 mg/day and 100 mg/day are assumed for CTE and RME exposure, respectively. The exposure frequency is estimated to be 19.5 days per year for CTE individuals and 100 days per year for RME individuals, based on the mean (39 visits per year for CTE) and 90th percentile (100 visits per year for RME) of visits to Jefferson County Open Space (Jefferson County Open Space Department, 1996) and the assumption that 50% (CTE) and 100% (RME) of all visits occur at the Richardson Flats site.

Exposure Parameter for Soil/Tailings Ingestion	CTE	RME
IR (kg/event)	50	100
BW (kg)	70	70
EF (events/year)	19.5	100
ED (years)	7	24
AT (non-cancer effects) (days)	7·365	24·365
AT (cancer effects) (days)	70·365	70·365

Based on these exposure parameters, the HIF values for exposure of high intensity recreational visitors to tailings and contaminated soil are as follows:

Recreational Exposure to Soil/Tailings	HIF (kg/kg-d)	
	CTE	RME
Chronic (non-cancer)	3.8E-08	3.9E-07
Lifetime (cancer)	3.8E-09	1.3E-07

Inhalation of Particulates

The basic equation recommended by USEPA (1989a) for evaluating exposure due to inhalation of a chemical in air is:

$$DIa = Ca \bullet \left(\frac{BR}{BW}\right) \bullet \left(\frac{ET \bullet EF \bullet ED}{AT}\right)$$

where:

DI_a = Daily Intake from inhalation of a chemical in air (mg/kg-d)
C_a = Concentration of chemical in air (mg/m^3)
BR = Breathing rate of air (m^3/hour)
ET = Exposure time (hours/day)
EF = Exposure frequency (days/year)
ED = Exposure duration (years)
BW = Body weight (kg)
AT = Averaging time (days)

Default values and assumptions recommended by USEPA (1989a, 1991b, 1993a) for evaluation of exposure to particulates in air are listed below. An inhalation rate of 2.4 m^3/hr for adults was based on the average of medium and heavy activity inhalation rates for this age group. This information is from the 1997 Exposure Factors Handbook and was used as inputs in the Rocky Flats Task 3 Report (USEPA, 2001b). The Exposure Time was based on the 1995 Boulder County open space survey (Boulder County Open Space Operations, 1995) of time spent on site (19% < 1 hour, 71% 1-3 hours, 9% 4-6 hours, and 1% >7 hours). Values of 1.5 and 2.5 hours/day were selected for the CTE and RME exposures, respectively. Although this information pertains to a different site, the values are judged to be applicable at Richardson Flats. The exposure frequency is estimated to be 19.5 days per year for CTE individuals and 100 days per year for RME individuals, based on the mean (39 visits per year for CTE) and 90th percentile (100 visits per year for RME) of visits to Jefferson County Open Space (Jefferson County Open Space Department, 1996) and the assumption that 50% (CTE) and 100% (RME) of all visits occur at the Richardson Flats site.

Exposure Parameters for Inhalation of Particulates	CTE	RME
BR (m^3/hr)	2.4	2.4
BW (kg)	70	70
ET (hr/day)	1.5	2.5
EF (days/yr)	19.5	100
ED (years)	7	24
AT (non-cancer effects) (days)	7·365	24·365
AT (cancer effects) (days)	70·365	70·365

Based on the exposure parameters above, the HIFs for exposure to particulates are as follows:

Recreational Exposure to Particulates	HIF (m^3/kg-d)	
	CTE	RME
Chronic (non-cancer)	2.74E-03	2.3E-02
Lifetime (cancer)	2.7E-04	8.1E-03

5.1.4 Concentration of Arsenic in Site Media

When people are exposed to a chemical in a medium such as soil, the level of exposure and risk is proportional to the average concentration in the area where exposure occurs. The location where exposure occurs (e.g., a specific residential yard or house) is usually referred to as the Exposure Unit (EU), and the average concentration within the EU is referred to as the Exposure Point Concentration (EPC). Typically, the EPC is estimated based on a set of measured values of the medium collected from the EU. However, the simple average of the measured values is only an estimate of the true mean, and the actual value could be either higher or lower. Because of this uncertainty, the USEPA typically recommends that, for chemicals such as arsenic, the EPC that is used to calculate exposure and risk be based on either the 95% upper confidence limit (UCL) of the mean concentration or the maximum concentration (whichever is lower) (USEPA 1989a). Note that this approach is used for both the CTE and the RME exposure scenarios (USEPA 1992a). The equation used to calculate the UCL depends on what is known about the underlying distribution of values. In most cases, it is assumed the distribution is right-skewed, and the equation for a lognormal distribution is used (USEPA 1992a). However, when the data are described by a distribution that is more nearly symmetric, then the equation for a t-distribution is used (USEPA 1992a). Samples that are below the detection limit are evaluated using a value equal to one-half the detection limit.

Arsenic concentrations in site media and EPCs are summarized below.

Media	Avg (ppm)	Min (ppm)	Max (ppm)	95th UCL (ppm)	EPC (ppm)
Sediment	162	101	310	200	200
Surface Water	0.008	0.003	0.75	0.012	0.012
Soil/Tailings	41	2.5	243	55	55

Although limited data on air concentrations are available for the site, these are too limited and were determined to be not suitable for use in the risk assessment (see Section 3.7). Therefore, arsenic concentrations in air were estimated using a simple emissions model (USEPA, 1996a):

$$Cair = Csoil * PEF$$

where:

C_{air} = Concentration of chemical in air (mg/m^3)
C_{soil} = Concentration of chemical in soil (mg/kg)
PEF = Particulate Emissions Factor (kg/m^3)

The PEF value depends on the local site conditions and on the nature of the force leading to soil suspension (i.e., wind or mechanical activity). Appendix E presents the derivation of these values. Estimated arsenic concentrations in air for low intensity and high intensity users are calculated as follows:

Release Mechanism	Exposed Population	Csoil (mg/kg)	PEF[a] (kg/m³)	Concentration (mg/m³)
Wind	Low Intensity User	55	2.92E-11	1.62E-09
Dirt Bike	High Intensity User	55	9.11E-08	5.05E-06

(a) See Appendix E for derivation

5.1.4 Relative Bioavailability (RBA)

Accurate assessment of the human health risks resulting from oral exposure to metals requires knowledge of the amount of metal absorbed from the gastrointestinal tract into the body. This information is especially important for environmental media such as soil or mine wastes, because metals in these media may exist, at least in part, in a variety of poorly water soluble minerals, and may also exist inside particles of inert matrix such as rock or slag. These chemical and physical properties may tend to influence (usually decrease) the absorption (bioavailability) of the metals when ingested.

At this site, no site-specific data are available for the bioavailability of arsenic in soils/tailings, therefore the Region 8 USEPA default value of 0.80 was utilized (USEPA, 1993b). For water, and RBA of 1.0 was assumed.

5.2 Toxicity Assessment

The toxic effects of a chemical generally depend not only upon the inherent toxicity of the compounds and the level of exposure (dose), but also on the route of exposure (oral, inhalation, dermal) and the duration of exposure (subchronic, chronic or lifetime). Thus, a full description of the toxic effects of a chemical includes a listing of what adverse health effects the chemical may cause, and how the occurrence of these effects depend upon dose, route, and duration of exposure.

The toxicity assessment process is usually divided into two parts: the first characterizes and quantifies the non-cancer effects of the chemical, while the second addresses the cancer effects of the chemical. This two-part approach is employed because there are typically major differences in the time-course of action and the shape of the dose-response curve for cancer and non-cancer effects.

Non-Cancer Effects

Essentially all chemicals can cause adverse health effects if given at a high enough dose. However, when the dose is sufficiently low, typically no adverse effect is observed. Thus, in characterizing the non-cancer effects of a chemical, the key parameter is the threshold dose at which an adverse effect first becomes evident. Doses below the threshold are considered to be safe, while doses above the threshold are likely to cause an effect.

The threshold dose is typically estimated from toxicological data (derived from studies of humans and/or animals) by finding the highest dose that does not produce an observable adverse effect, and the lowest dose which does produce an effect. These are referred to as the "No-observed-adverse-effect-level" (NOAEL) and the "Lowest-observed-adverse-effect-level" (LOAEL), respectively. The threshold is presumed to lie in the interval between the NOAEL and the LOAEL. However, in order to be conservative (protective), non-cancer risk evaluations are not based directly on the threshold exposure level, but on a value referred to as the Reference Dose (RfD). The RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime.

The RfD is derived from the NOAEL (or the LOAEL if a reliable NOAEL is not available) by dividing by an "uncertainty factor". If the data are from studies in humans, and if the observations are considered to be very reliable, the uncertainty factor may be as small as 1.0. However, the uncertainty factor is normally at least 10, and can be much higher if the data are limited. The effect of dividing the NOAEL or the LOAEL by an uncertainty factor is to ensure that the RfD is not higher than the threshold level for adverse effects. Thus, there is always a "margin of safety" built into an RfD, and doses equal to or less than the RfD are nearly certain to be without any risk of adverse effect. Doses higher than the RfD may carry some risk, but because of the margin of safety, a dose above the RfD does not mean that an effect will necessarily occur.

Cancer Effects

For cancer effects, the toxicity assessment process has two components. The first is a qualitative evaluation of the weight of evidence that the chemical does or does not cause cancer in humans. Typically, this evaluation is performed by the USEPA, using the system summarized in the table below:

Category	Meaning	Description
A	Known human carcinogen	Sufficient evidence of cancer in humans.
B1	Probable human carcinogen	Suggestive evidence of cancer incidence in humans.
B2	Probable human carcinogen	Sufficient evidence of cancer in animals, but lack of data or insufficient data from humans.
C	Possible human carcinogen	Suggestive evidence of carcinogenicity in animals.
D	Cannot be evaluated	No evidence or inadequate evidence of cancer in animals or humans.

For chemicals which are classified in Group A, B1, B2, or C, the second part of the toxicity assessment is to describe the carcinogenic potency of the chemical. This is done by quantifying how the number of cancers observed in exposed animals or humans increases as the dose increases. Typically, it is assumed that the dose response curve for cancer has no threshold, arising from the origin and increasing linearly until high doses are reached. Thus, the most convenient descriptor of cancer potency is the slope of the dose-response curve at low dose (where the slope is still linear). This is referred to as the Slope Factor (SF), which has dimensions of risk of cancer per unit dose.

Estimating the cancer Slope Factor is often complicated by the fact that observable increases in cancer incidence usually occur only at relatively high doses, frequently in the part of the dose-response curve that is no longer linear. Thus, it is necessary to use mathematical models to extrapolate from the observed high dose data to the desired (but unmeasurable) slope at low dose. In order to account for the uncertainty in this extrapolation process, USEPA typically chooses to employ the upper 95th confidence limit of the slope as the Slope Factor. That is, there is a 95% probability that the true cancer potency is lower than the value chosen for the Slope Factor. This approach ensures that there is a margin of safety in cancer risk estimates.

5.2.1 Adverse Effects of Arsenic

Excess exposure to arsenic is known to cause a variety of adverse health effects in humans. These effects depend on exposure level (dose) and also on exposure duration. The following sections discuss the most characteristic of these effects.

Noncancer Effects

Oral exposure to high doses of arsenic produces marked acute irritation of the gastrointestinal tract, leading to nausea and vomiting. Symptoms of chronic ingestion of lower levels of arsenic often begin with a vague weakness and nausea. As exposure continues, symptoms become more characteristic and include diarrhea, vomiting, decreased blood cell formation, injury to blood vessels, damage to kidney and liver, and impaired nerve function that leads to "pins and needles" sensations in the hands and feet. The most diagnostic sign of chronic arsenic exposure is an unusual pattern of skin abnormalities, including dark and white spots and a pattern of small "corns," especially on the palms and soles (ATSDR 1991).

The long-term (chronic) average daily intake of arsenic that produces these effects varies from person to person. In a large epidemiological study, Tseng et al. (1968) reported skin and vascular lesions in humans exposed to 1.4E-02 mg/kg/day or more arsenic through drinking water in Taiwan. These effects were not observed in a control population ingesting 8.0E-04 mg/kg/day. Based on this, the USEPA calculated a chronic oral reference dose (RfD) of 3.0E-04 mg/kg/day (IRIS, 1998). This is a dose which is believed to be without significant risk of causing adverse noncancer effects in even the most susceptible humans following chronic exposure.

Carcinogenic Effects

There have been a number of epidemiological studies in humans which indicate that chronic inhalation exposure to arsenic is associated with increased risk of lung cancer (USEPA 1984, ATSDR 1991). In addition, there is strong evidence from a number of human studies that oral exposure to arsenic increases the risk of skin cancer (USEPA 1984, ATSDR 1991). The most common type of cancer is squamous cell carcinoma, which appears to develop from some skin corns. In addition, basal cell carcinoma may also occur, typically arising from cells not associated with the corns. Although these cancers may be easily removed, they can be painful and disfiguring and can be fatal if left untreated. Although the evidence is limited, there are some reports which indicate that chronic oral arsenic exposure may also increase risk of internal cancers, including cancer of the liver, bladder and lung, and that inhalation exposure may also increase risk of gastrointestinal, renal or bladder cancers (ATSDR 1991). Based on these data, USEPA has assigned arsenic to cancer weight of evidence Category A.

The amount of arsenic ingestion that leads to skin cancer is controversial. Based on a study of skin cancer incidence in Taiwanese residents exposed mostly to As(+3) in drinking water (Tseng et al. 1968, USEPA 1984), the USEPA has calculated a unit risk of 5E-05 (ug/L)-1 corresponding to an oral slope factor of 1.5E+00 (mg/kg/day)-1 (IRIS 1998). This study has been criticized on several grounds, including uncertainty about exposure levels, possible effects of poor nutrition in the exposed population, potential exposure to other substances besides arsenic, and lack of blinding in the examiners. Consequently, some quantitative uncertainty exists in the cancer potency factor derived from the Tseng data. Nevertheless, these criticisms do not challenge the fundamental conclusion that arsenic ingestion is associated with increased risk of skin cancer, and the Tseng study is considered to be the best study currently available for quantitative estimation of skin cancer risk.

There are good data to show that arsenic is metabolized by methylation in the body, and some researchers have suggested that this could lead to a threshold dose below which cancer will not occur. Although there are data which are consistent with this view, the USEPA has reviewed the available information (USEPA 1988) and has concluded that the data are insufficient at present to establish that there is a threshold for arsenic-induced cancer.

5.2.2 Summary of Oral Toxicity Values

The toxicity factors derived by the USEPA for oral exposure to the site COPCs are summarized below:

Chemical	Non-Cancer RfD (mg/kg-day)	Cancer	
		Weight-of-Evidence	oral SF $(mg/kg\text{-}day)^{-1}$
Arsenic	3E-04	A	1.5

5.3 Risk Characterization

5.3.1 Overview

Risk characterization is the process of combining information on doses (Section 5.1) with toxicity information (Section 5.2) in order to estimate the nature and likelihood of adverse effects occurring in members of the exposed population. As explained earlier, this process is usually performed in two steps, the first addressing noncancer risks from chemicals of concern, and the second addressing cancer risks. The basic methods used to quantify noncancer and cancer risks are summarized below.

5.3.2 Noncancer Risk

Basic Equations

The potential for noncancer effects from exposure to a chemical is evaluated by comparing the estimated daily intake of the chemical over a specific time period with the RfD for that chemical derived for a similar exposed period. This comparison results in a noncancer Hazard Quotient, as follows (USEPA 1989a):

$$HQ = DI/RfD$$

where:

HQ = Hazard Quotient
DI = Daily Intake (mg/kg-day)
RfD = Reference Dose (mg/kg-day)

If the HQ for a chemical is equal to or less than one (1E+00), it is believed that there is no appreciable risk that noncancer health effects will occur. If an HQ exceeds 1E+00, there is some possibility that noncancer effects may occur, although an HQ above 1E+00 does not indicate an effect will definitely occur. This is because of the margin of safety inherent in the derivation of all RfD values. However, the larger the HQ value, the more likely it is that an adverse effect may occur. If more than one chemical affects the same target tissue or organ system (e.g., the liver), then the total risk of adverse effects in that tissue is referred to as the Hazard Index (HI), and is estimated by summing the HQ values for all chemicals that act on that tissue.

5.3.3 Cancer Risk

Basic Equations

The risk of cancer from exposure to a chemical is described in terms of the probability that an exposed individual will develop cancer because of that exposure by age 70. For each chemical of concern, this value is calculated from the daily intake of the chemical from the site, averaged over a lifetime (DI_L), and the SF for the chemical, as follows (USEPA 1989a):

$$\text{Cancer Risk} = 1 - \exp(-DI_L \times SF)$$

In most cases (except when the product of DI_L*SF is larger than about 0.01), this equation may be accurately approximated by the following:

$$\text{Cancer Risk} = DI_L \times SF$$

The level of cancer risk that is of concern is a matter of individual, community and regulatory judgement. However, the USEPA typically considers risks below 1E-06 to be so small as to be negligible, and risks above 1E-04 to be sufficiently large that some sort of action or intervention is usually needed (USEPA, 1991b). Risks between 1E-04 and 1E-06 usually do not require action (USEPA, 1991b), but this is evaluated on a case by case basis.

5.3.4 Results

Non-Cancer Risks

The following table summarize the estimated HQ values for both low and high intensity recreational visitors exposed to arsenic in site media. As shown, none of the media exceeds an HQ of 1E+00 for either low or high intensity use scenarios for either average or RME exposure conditions. The majority of observed risk is attributable to soil/tailings ingestion.

Population	Exposure Pathway	Average	RME
Low Intensity	Sediment Ingestion	2E-03	1E-02
	Surface Water Ingestion	2E-05	9E-04
	Dermal Contact with Surface Water	1E-05	2E-04
	Soil/Tailings Ingestion	8E-03	8E-02
	Inhalation of Particulates in Air	2E-08	1E-07
High Intensity	Soil/Tailings Ingestion	6E-03	6E-02
	Inhalation of Particulates in Air	4E-05	3E-04
Total Risk Low Intensity User		*1E-02*	*9E-02*
Total Risk High Intensity User		*6E-03*	*6E-02*

Cancer Risks

Using these equations, the estimated lifetime average and RME daily intake values (calculated as described in Section 5.1) for both low and high intensity users were combined with the oral slope factor for arsenic discussed in Section 5.2. The detailed calculations are presented in Appendix F, and the results are summarized in the following table. As seen, the majority of observed risk is attributable to soil/tailing ingestion. However, total cancer risks do not exceed a level of 1E-04 for low intensity and high intensity users using either average or RME exposure assumptions.

Population	Exposure Pathway	Average	RME
Low Intensity	Sediment Ingestion	1E-07	3E-06
	Surface Water Ingestion	8E-10	2E-07
	Dermal Contact with Surface Water	7E-10	3E-08
	Soil/Tailings Ingestion	6E-07	2E-05
	Inhalation of Particulates in Air	1E-11	3E-10
High Intensity	Soil/Tailings Ingestion	3E-07	1E-05
	Inhalation of Particulates in Air	2E-08	6E-07
Total Risk Low Intensity User		*7E-07*	*2E-05*
Total Risk High Intensity User		*3E-07*	*1E-05*

5.4 Uncertainties

It is important to recognize that the exposure and risk calculations for the COPCs presented in this section are based on a number of assumptions, and that these assumptions introduce uncertainty into the dose and risk estimates. Assumptions are required because of data gaps in our understanding of the toxicity of chemicals, and in our ability to estimate the true level of human exposure to chemicals. In most cases, assumptions employed in the risk assessment process to deal with uncertainties are intentionally conservative; that is, they are more likely to lead to an overestimate than an underestimate of risk. It is important for risk managers and the public to take these uncertainties into account when interpreting the risk conclusions derived for this site.

5.4.1 Uncertainties in Concentration Estimates

Evaluation of human health risk at any particular location requires accurate information on the average concentration level of a COPC at that location. However, concentration values may vary from sample to sample, so the USEPA recommends that the 95% upper confidence limit of the mean be used in evaluation of both average and RME exposure and risk. This approach typically ensures that all of the risk estimates are more likely to be high than low.

Risks from exposure to non-lead COPCs were evaluated based on surficial soil data. This decision was based on the assumptions that recreational users be most likely to be exposed to surficial soils based on their activities. If the depth distribution for arsenic mimics that observed for lead, risks from exposure to subsurface soils will be similar or less than those observed for surface soils. However, if concentrations for these analytes are found to increase as a function of depth, the risks based on surface soil exposure will underestimate risks for those individuals exposed to buried materials. A quick review of the data show that the maximum arsenic concentration in soil/tailings observed at the site at any depth is 637 mg/kg. Using this value in the risk calculations, total non-cancer risks to the low and high intensity recreational user are 9E-01and 7E-01, respectively. Cancer risks 2E-04 and 1E-04, respectively.

5.4.2 Uncertainties in Human Intake

As discussed in Section 5.1, there is usually wide variation between different individuals with respect to the level of contact they may have to chemicals in the environment. This introduces uncertainty into the most appropriate values to use for exposure parameters such as soil and dust intake rates, number of years at the residence, etc. Because of the uncertainty in the most appropriate values for these parameters, the USEPA generally recommends default values that are more likely to overestimate than underestimate exposure and risk.

Additionally, in the absence of default values or site-specific information on the intake rates for recreational visitors, intake rates were estimated or approximated based on existing guidance, information from other sites and based on professional judgement. For soil/tailings and sediment ingestion, the intake rates for recreational users are extrapolated from the recommended default values for residential incidental ingestion of soil. For water, intake rates for ingestion during non-immersion contact activities (wading) are extrapolated from USEPA default values for immersion contact (swimming) activities. These assumptions and extrapolations are conservative, and thus more likely to overestimate than underestimate exposure and risk.

5.4.3 Uncertainties in Toxicity Values

One of the most important sources of uncertainty in a risk assessment is in the RfD values used to evaluate noncancer risk and in the slope factors used to quantify cancer risk. In many cases, these values are derived from a limited toxicity database, and this can result in substantial uncertainty, both quantitatively and qualitatively. For example, there is continuing scientific debate on the accuracy of the oral slope factor and the oral Reference Dose for arsenic and whether or not they are accurate and appropriate for predicting hazards from relatively low dose exposures. In order to account for these and other uncertainties associated with the evaluation of toxicity data, both RfDs and SFs are derived by the USEPA in a way that is intentionally conservative; that is, risk estimates based on these RfDs and SFs are more likely to be high than low.

5.4.4 Uncertainties in Absorption from Soil

Another important source of uncertainty regarding the toxicity of arsenic is the degree to which it is absorbed into the body after ingestion of soil. Toxicity factors (RfD, oSF) for arsenic are based on observed dose response relationships when exposure occurs by ingestion of arsenic dissolved in water. If arsenic in soil is not absorbed as well as arsenic in water, use of unadjusted toxicity factors will tend to overestimate risk. At this site, the USEPA default relative bioavailability factor for arsenic of 0.8 was used for soil/tailings and sediment. However, use of this factor may or may not be reflective of the actual site RBA. Tests in juvenile swine have shown that RBA values in site soils may be higher or lower than the default value based on soil characteristics such as mineral phase, particle size distribution, etc.

Site specific studies of arsenic bioavailability in mining wastes and soils conducted throughout Region 8 (e.g., California Gulch, Clark Fork, and Murray Smelter) suggest that actual site RBAs can be lower than the USEPA default value. For these sites, the arsenic RBA in soil or mining waste materials ranged from 0.14 to 0.57. If the bioavailability of arsenic in soil and tailing at the Richardson Flats site is similar to the arsenic RBA reported at other mining sites, the total risk from arsenic at the site would be lower. For example, substituting an arsenic RBA of 0.4 for the USEPA default would result in a decrease in the risk from arsenic at the site by a factor of 2.

5.4.5 Uncertainties from Pathways Not Evaluated

As discussed in Section 4, not all possible pathways of human exposure to site COPCs were evaluated quantitatively in this risk assessment, and omission of these pathways presumably leads to some degree of underestimation of total risk. For some of these pathways (dermal absorption from soil on the skin), the underestimation of risk is believed to be minimal (see Appendix D). In the case of ingestion of site biota, the magnitude of the underestimation is less certain. Studies at other sites (Sverdrup, 1995) suggest that exposure by this pathways is probably not as large as by oral exposure, but that the contribution is not completely negligible. However, the magnitude of this risk contributed by pathway is expected to vary widely from site to site, depending on the amount of uptake from soil into the biota and the amount and type of biota actually consumed by site visitors. At this time, it is not thought that this pathway is a prevalent pathway of exposure to area visitors.

5.4.6 Uncertainties in Summing Risks Across Exposure Pathways

In accord with USEPA guidance (1989a), risks from each exposure pathway that apply to the same exposed individual are summed to estimate the total risk to that individual. In the case of CTE receptors, summation of CTE risks across different exposure pathways is likely to yield a reasonable estimate of total risk. In the case of RME receptors, summation of RME risks across different pathways that are independent of each other may tend to be conservative, since the same individual may not be at the high end of the exposure distribution for all pathways. For example, at this site, a low intensity recreational visitor may not simultaneously experience RME exposures from soil/tailing and from surface water and sediments. Thus, summation of RME risks across different (and independent) exposure pathways should be viewed as a conservative screening-level approach for estimation of total risk.

6.0 RISKS FROM LEAD

As noted earlier, risks from lead are evaluated using a somewhat different approach than for most other metals. First, because lead is widespread in the environment, exposure can occur by many different pathways. Thus, lead risks are usually based on consideration of total exposure (all pathways) rather than just to site-related exposures. Second, because studies of lead exposures and resultant health effects in humans have traditionally been described in terms of blood lead level (PbB, expressed in units of ug/dL), lead exposures and risks are typically assessed using an uptake-biokinetic model rather than an RfD approach. Therefore, calculating the level of exposure and risk from lead in soil also requires assumptions about the level of lead in other media, and also requires use of pharmacokinetic parameters and assumptions that are not needed in traditional methods.

For residential land use, the sub-population of chief concern is young children. This is because young children 1) tend to have higher exposures to lead in soil, dust and paint, 2) tend to have a higher absorption fraction for ingested lead, and 3) are more sensitive to the toxic effects of lead than are older children or adults. For non-residential exposures (e.g., recreation, occupational) the population of chief concern are older children and young adults. When adults are exposed, the sub-population of chief concern is pregnant women and women of child-bearing age, since the blood lead level of a fetus is nearly equal to the blood lead level of the mother (Goyer 1990).

At this site, the BHHRA focuses on risks to recreational visitors. For low-intensity users, the visitors were assumed to range from young children to adults, whereas high-intensity visitors were assumed to be teenagers and adults. Because the effects of lead exposure are evaluated differently for young children than they are for adults, two separate modeling approaches were used to evaluate risks to the recreational visitors: one specific to children (low-intensity only) and one appropriate for older individuals (low- and high-intensity). These approaches are described in further detail below (Section 6.2).

6.1 Adverse Effects of Lead Exposure

Excess exposure to lead can result in a wide variety of adverse effects in humans. Chronic low-level exposure is usually of greater concern for young children than older children or adults. There are several reasons for this focus on young children, including the following: 1) young children typically have higher exposures to lead-contaminated media per unit body weight than adults, 2) young children typically have higher lead absorption rates than adults, and 3) young children are more susceptible to effects of lead than are adults. The following sections summarize the most characteristic and significant of the adverse effects of lead on children, and current guidelines for classifying exposures as acceptable or unacceptable.

6.1.1 Neurological Effects

The effect of lead that is usually considered to be of greatest concern in children is impairment of the nervous system. Many studies have shown that animals and humans are most sensitive to the effects of lead during the time of nervous system development, and because of this, the fetus, infants and young children (0-6 years of age) are particularly vulnerable. The effects of chronic low-level exposure on the nervous system are subtle, and normally cannot be detected in individuals, but only in studies of groups of children. Common measurement endpoints include various types of tests of intelligence, attention span, hand-eye coordination, etc. Most studies observe effects in such tests at blood lead levels of 20-30 ug/dL, and some report effects at levels as low as 10 ug/dL and even lower. Such effects on the nervous system are long-lasting and may be permanent.

6.1.2 *Effects on Pregnancy and Fetal Development*

Studies in animals reveal that high blood lead levels during pregnancy can cause fetotoxic and teratogenic effects. Some epidemiologic studies in humans have detected an association between elevated blood lead levels and endpoints such as decreased fetal size or weight, shortened gestation period, decreased birth weight, congenital abnormalities, spontaneous abortion and stillbirth (USEPA 1986). However, these effects are not detected consistently in different studies, and some researchers have detected no significant association between blood lead levels and signs of fetotoxicity. On balance, these data provide suggestive evidence that blood lead levels in the range of 10-15 ug/dL may cause small increases in the risk of undesirable prenatal as well as postnatal effects, but the evidence is not definitive.

6.1.3 *Effects on Heme Synthesis*

A characteristic effect of chronic high lead exposure is anemia stemming from lead-induced inhibition of heme synthesis and a decrease in red blood cell life span. ACGIH (1995) concluded that decreases in ALA-D activity (a key early enzyme involved in heme synthesis) can be detected at blood lead levels below 10 ug/dL. Heme synthesis is inhibited not only in red blood cells but in other tissues. Several key enzymes that contain heme, including those needed to form vitamin D, also show decreased activity following lead exposure (USEPA 1986). The Centers for Disease Control (CDC 1991) reviewed studies on the synthesis of an active metabolite of vitamin D and found that impairment was detectable at blood lead levels of 10 - 15 ug/dL.

6.1.4 *Cancer Effects*

Studies in animals indicate that chronic oral exposure to very high doses of lead salts may cause an increased frequency of tumors of the kidney (USEPA 1989b, ACGIH 1995). However, there is only limited evidence suggesting that lead may be carcinogenic in humans, and the noncarcinogenic effects on the nervous system are usually considered to be the most important and sensitive endpoints of lead toxicity (USEPA 1988). ACGIH (1995) states that there is insufficient evidence to classify lead as a human carcinogen.

6.1.5 *Current Guidelines for Protecting Children from Lead*

It is currently difficult to identify what degree of lead exposure, if any, can be considered safe for infants and children. As discussed above, some studies report subtle signs of lead-induced effects in children and perhaps adults beginning at around 10 ug/dL or even lower, with population effects becoming clearer and more definite in the range of 30-40 ug/dL. Of special concern are the claims by some researchers that effects of lead on neurobehavioral performance, heme synthesis, and fetal development may not have a threshold value, and that the effects are long-lasting (USEPA 1986). On the other hand, some researchers and clinicians believe the effects that occur in children at low blood lead levels are so minor that they need not be cause for concern.

After a thorough review of all the data, the USEPA identified 10 ug/dL as the concentration level at which effects begin to occur that warrant avoidance, and has set as a goal that there should be no more than a 5% chance that a child will have a blood lead value above 10 ug/dL (USEPA, 1991b). Likewise, the Centers for Disease Control (CDC) has established a guideline of 10 ug/dL in preschool children which is believed to prevent or minimize lead-associated cognitive deficits (CDC 1991).

6.2 Evaluation of Lead Risks to Recreational Visitors

6.2.1 Evaluation of Lead Risks to Recreational Children

The standard model developed by the USEPA to assess the risks of lead exposure in children is referred to as the Integrated Exposure Uptake and Biokinetic (IEUBK) model. This model requires as input data on the levels of lead in various environmental media at a specific location, and on the amount of these media contacted by a child living at that location. The inputs to the IEUBK model are selected to reflect estimates of central tendency values (i.e., arithmetic means or medians). These estimated inputs are used to calculate an estimate of the central tendency (the geometric mean) of the distribution of blood lead values that might occur in a population of children exposed to the specified conditions. Assuming the distribution is lognormal, and given (as input) an estimate of the variability between different children (this is specified by the geometric standard deviation or GSD), the model calculates the expected distribution of blood lead values, and estimates the probability that any random child might have a blood lead value over 10 ug/dL.

For this site, two simulations were run using the IEUBK model. The first evaluated risks to a hypothetical nearby resident. The second simulation was used to address the risk observed when the hypothetical residential child engaged in low-intensity recreational activities at the site. By comparing the two simulations and resulting predictions of blood lead concentrations, the excess risk attributable to the low-intensity recreational exposure can be identified.

A detailed printout of the input values used to evaluate lead risks for each scenario is presented in Appendix G. The following sections summarize the input parameters used for these calculations.

<u>Lead Concentration in Soil/Tailings and Intake Assumptions</u>

As discussed previously (Section 3.2.2), background soils were collected from areas surrounding the site. Although the samples do not represent "pristine" (not influenced by human activity) environmental levels, they are thought to be adequate to serve as a potential "off-site" residential concentration. Therefore, these background data were compiled and a value of 64 mg/kg of lead in soil, representing the log-normal UCL95 value was utilized for residential exposure. Indoor dust concentrations were calculated using the USEPA default ($C_{dust} = 0.7 * C_{yard\ soil}$). Other intake parameters for the residential scenario were kept as IEUBK model defaults.

The second scenario combined the residential parameters with those for occasional recreational visits. These visitor parameters were based on the average child who is thought to engage in recreational activities at the site 19.5 days/year (39 recreational visits (days) per year * 50%of total visits at the Richardson Flats Site) and consume 50 mg of soil during each recreational event. Because recreational activities are not thought to occur 365 days/year, a time-weighted approach was used to derive values for input into the IEUBK model. Therefore, if the child visited the site 19.5 days/year they were exposed to their soil intake at the site on those days. For the remaining 315 days/year the child was assumed to be exposed at home at the concentration specified above. The concentration utilized for recreational exposure was the log-normal UCL95 of the surficial on-site soil and tailings, which was determined to be 1,331 mg/kg. The following table summarizes both intake and concentration parameters for soil/tailings. The weighted average value shows the number input into the IEUBK model for the combined residential/recreational exposure scenario.

Age	Scenario	Days/Year	Intake (mg/day)	Concentration (mg/kg)
0-1	Residential	345.5	85	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**83**	**105**
1-2	Residential	345.5	135	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**130**	**90**
2-3	Residential	345.5	135	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**130**	**90**
3-4	Residential	345.5	135	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**130**	**90**
4-5	Residential	345.5	100	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**97**	**99**
5-6	Residential	345.5	90	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**88**	**103**
6-7	Residential	345.5	85	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**83**	**105**

Water and Air

For this analysis, lead concentrations in water and intake assumptions for each scenario were calculated according to the approach used above for soil/tailings. Residential water concentrations and intakes were set equal to the IEUBK default values. Because the intake rates (5 mL/event) and the site-specific lead concentrations (0.07 ug/L) are so low, the calculated weighted average was the same for the combined residential/recreational scenario as for the residential alone. Therefore, these values were the same in both model simulations.

Lead values for air were kept at the IEUBK default value of 0.1 ug/m^3. This is based on the observation that the maximum lead concentrations in soil/tailing (5,875 mg/kg) would result in a predicted air concentration of

0.007 ug/m^3 using a PEF of 1.16E-9 kg/m3 for low intensity activities. Because this number was lower than the default value, the default was retained in the IEUBK model.

Diet

The default values of lead intake from the diet in the IEUBK model are based on dietary data from 1982 - 1988. Recent FDA data provide strong evidence that concentrations of lead in food have continued to decline since 1988. Based on interpretations of the data, and an extrapolation from the downward trend observed in the 1980's, it has been estimated that the average lead intake from food by children has declined by approximately 30% (Griffin et al., 1999a). Therefore the dietary values were obtained by multiplying the model default values by a factor of 0.70. The resulting values are presented below:

Age (years)	Adjusted Dietary Intake (ug/day)
0-1	3.87
1-2	4.05
2-3	4.54
3-4	4.37
4-5	4.21
5-6	4.44
6-7	4.90

Other

Recreational visitors are thought to be exposed to sediments at the site an average of 2 times/year while visiting the site. During each visit, children are assumed to ingest 25 mg of sediment. Based on a log-normal 95UCL lead concentration of 4,446 mg/kg in sediments, this is expected to result in an additional 0.61 ug/day of lead on a yearly basis. Therefore, in the combined residential/recreational scenario, a value of 0.61 ug/day of lead intake from other media was added for all age groups (0 to 6 years).

Age

Predicted blood lead values were calculated for each scenario (residential & residential + recreational) for a child 0-84 months of age.

Absorption Fraction for Lead in Soil and Sediment

The absorption fraction is a measure of the amount of metal absorbed from the gastrointestinal tract into the body. This information is especially important for environmental media such as soil or mine wastes, because metals in these media may exist, at least in part, in a variety of poorly water soluble minerals, and may also exist inside particles of inert matrix such as rock or slag. These chemical and physical properties may tend to influence (usually decrease) the absorption (bioavailability) of the metals when ingested. Because no site specific data on bioavailability were available at this site, the default value of 0.60 was used in the model.

GSD

The GSD recommended as the default for the IEUBK model is 1.6 (USEPA 1994b and 1994c). However, several blood lead studies that have been performed in the Salt Lake City area have yielded GSD estimates of about 1.4 (Griffin et al., 1999b). Therefore, values of both 1.6 and 1.4 were evaluated in this assessment.

Results

Using the input parameters identified above, geometric mean blood lead values and P10 values were calculated for both scenarios using the IEUBK model (IEUBKwin32 build 250). The results are summarized below:

Scenario	GSD = 1.4		GSD = 1.6	
	Geometric Mean Blood Lead (ug/dL)	P10	Geometric Mean Blood Lead (ug/dL)	P10
Residential Only	1.8	<0.01%	1.8	0.01%
Residential + Low Intensity Recreational	2.0	<0.01%	2.0	0.01%

As seen, children who engage in low-intensity recreational activities at this site have higher predicted blood lead levels than those with no recreational exposure. However, the geometric mean values are relatively low and children engaging in recreational activities have under a 5% chance of exceeding a blood lead value of 10 ug/dL using a GSD value of either 1.4 or 1.6.

Based on the results of the IEUBK model, it is considered unlikely that low-intensity recreational exposures to lead in soil/tailings at this site will result in an elevation in blood lead levels which will exceed USEPA's guidelines of no more than a 5% chance that a child will have a blood lead value above 10 ug/dL .

6.2.2 Evaluation of Lead Risks to Recreational Teenagers and Adults

The IEUBK model developed by USEPA is intended for evaluation of lead risks to residential children, and is not appropriate for evaluation of lead risks to older children or adults exposed during either low- or high-intensity recreational activities. However, there are several mathematical models which have been proposed for evaluating lead exposure in adults, including those developed by Bowers et al. (1994), O'Flaherty (1993), Leggett (1993), and the State of California (CEPA 1992). Of these, the biokinetic slope factor approach described by Bowers et al. has been identified by USEPA's Technical Workgroup for Lead (USEPA 1996b) as a reasonable interim methodology for assessing risks to adults from exposure to lead and for establishing risk-based concentration goals that will protect older children and adults from lead. For this reason, this method was used for estimating risks from soil lead and tailings exposure that could be of concern to older children and adults at this site.

Basic Equation

The Bowers model predicts the blood lead level in an adult exposed to lead in a specified occupational setting by summing the "baseline" blood lead level (PbB_0) (that which would occur in the absence of any above-average site-related exposures) with the increment in blood lead that is expected as a result of increased exposure due to contact with a lead-contaminated site medium. The latter is estimated by multiplying the absorbed dose of lead from site-related exposure by a "biokinetic slope factor" (BKSF). Thus, the basic equation is:

$$PbB = PbB_0 + (PbS \cdot BKSF \cdot IR_s \cdot AF_s \cdot EF_s)/AT$$

where:

PbB = Central estimate of blood lead concentrations (ug/dL) in adults (i.e., women of child-bearing age) that have site exposures to soil lead at concentration, PbS.

PbB_0 = Typical blood lead concentration (ug/dL) in adults (i.e., women of child-bearing age) in the absence of exposures to the site that is being assessed.

BKSF = Biokinetic slope factor relating (quasi-steady state) increase in typical adult blood lead concentration to average daily lead uptake (ug/dL blood lead increase per ug/day lead uptake)

PbS = Soil lead concentration (ug/g) (appropriate average concentration for individual)

IR_s = Intake rate of soil, including both outdoor soil and indoor soil-derived dust (g/day)

AF_s = Absolute gastrointestinal absorption fraction for ingested lead in soil and lead in dust derived from soil (dimensionless). The value of AF_s is given by:

$$AF_s = AF(food) * RBA(soil)$$

EF_s = Exposure frequency for contact with assessed soils (days of exposure during the averaging period)

AT = Averaging time; the total period during which soil contact may occur; 365 days/year for continuing long term exposures.

Once the geometric mean blood lead value is calculated, the full distribution of likely blood lead values in the population of exposed people can then be estimated by assuming the distribution is lognormal with some specified geometric standard deviation (GSD). Specifically, the 95th percentile of the predicted distribution is given by the following equation (Aitchison and Brown 1957):

$$95th = GM \cdot GSD^{1.645}$$

Input values selected for each of these parameters are summarized below:

Parameter	Low Intensity User	High Intensity User	Source
PbB_0 (ug/dL)	1.36	1.36	USEPA (2002b, Table 3c) weighted average of females age 17- 45 years in the West Census Region.
PbS (ppm)	1331	1331	UCL95 Site lead concentration based on a log-normal distribution
BKSF (ug/dL per ug/day)	0.4	0.4	USEPA (1996b)
IR (g/day exposed)	0.025	0.05	Based on intake rate of 25 and 50 mg/day for low and high intensity users, respectively as discussed in Section 5. Multiplied by a factor of 1E-03 g/mg.
EF_s (days exposed at site/yr)	19.5	19.5	Based on CTE exposure assumptions for arsenic (see Section 5.1.2)
AT (days)	365	365	USEPA (1996b)
AF_o (unitless)	0.12	0.12	Based on an absorption factor for soluble lead of 0.20 (USEPA 1996b) and a relative bioavailability of 0.6
GSD	2.07	2.07	USEPA (2002, Table 3c) weighted average of females age 17- 45 years in the West Census Region.

Results

Based on these input parameters, the predicted geometric mean blood lead and PbB_{95} values for recreational visitors were calculated. For low intensity visitors, the geometric mean blood lead concentration was predicted to be 1.4 ug/dL with a PbB_{95} value of 4.8 ug/dL. In other words, it is predicted that 95% of the low intensity visitors will have a blood lead value less than 4.8 ug/dL. For high intensity visitors, the geometric mean blood lead concentration was predicted to be 1.5 ug/dL with a PbB_{95} value of 5.1 ug/dL. In other words, it is predicted that 95% of the high intensity visitors will have a blood lead value less than 5.1 ug/dL.

The USEPA has not yet issued formal guidance on the blood lead level that is considered appropriate for protecting the health of pregnant women or other adults. However, as noted above, USEPA recommends that there should be no more than a 5% likelihood that a young child should have a PbB value greater than 10 ug/dL (USEPA, 1991b). This same blood lead level (10 ug/dL) is also taken to be the appropriate goal for blood lead levels in the fetus, and hence in pregnant women and women of child-bearing age. Therefore, the health criterion selected for use in this evaluation is that there should be no more than a 5% chance that the blood level of a fetus will be above 10 ug/dL.

This health goal is equivalent to specifying that the 95th percentile of the PbB distribution in fetuses does not exceed 10 ug/dL:

$$PbB_{95}fetal \leq 10 \text{ ug/dL}$$

The relationship between fetal and maternal blood lead concentration has been investigated in a number of studies. Goyer (1990) reviewed a number of these studies, and concluded that there was no significant placental/fetal barrier for lead, with fetal blood lead values being equal to or just slightly less than maternal blood lead values. The mean ratio of fetal PbB to maternal PbB in three recent studies cited by Goyer was 0.90. Based on this, the 95th percentile PbB in the mother is then:

$$PbB_{95}maternal = 10/0.90 = 11.1 \text{ ug/dL.}$$

That is, the target blood lead level for pregnant women is estimated to be 11.1 ug/dL. Because individuals in the recreational population are assumed to be mainly age 12-49, it is possible that women of child-bearing age may also be included in this group, so the same target blood lead value is assumed to apply to this population as well.

A comparison of the 95^{th} percentile blood lead levels predicted for site recreational visitors shows that recreational use at this site is not predicted to result in blood lead levels which exceed a target concentration of 11.1 ug/dL.

6.3 Uncertainties

It is important to recognize that the exposure and risk calculations presented in this document are based on a number of assumptions, and that these assumptions introduce uncertainty into the exposure and risk estimates. Assumptions are required because of data gaps in our understanding of the toxicity of chemicals, and in our ability to estimate the true level of human exposure to chemicals. In most cases, assumptions employed in the risk assessment process to deal with uncertainties are intentionally conservative; that is, they are more likely to lead to an overestimate rather than an underestimate of risk. It is important for risk managers and the public to take these uncertainties into account when interpreting the risk conclusions derived for this site.

6.3.1 Uncertainty in Lead Concentration Estimates

Evaluation of human health risk at any particular location requires accurate information on the average concentration level of a COPC at that location. When the exposure area is small (e.g., a residential yard), use of the average concentration of lead in soil is appropriate(USEPA, 1994a). However, at the Richardson Flats Site the exposure area is large. Because estimating the mean is more difficult when aggregating data over a large exposure area and could underestimate the true mean, the 95^{th} UCL soil lead concentration was used to evaluate risks from lead. This approach is reasonable for use at the Richardson Flats site where lead concentrations in onsite soil/tailing materials range from 14 to 5,875 mg/kg. This conservative approach for estimating exposure to lead at the site may overestimate the actual risks from lead for the site, ensuring that all of the risk estimates are more likely to be high than low.

Risks from exposure to lead were evaluated based on surficial soil data. This decision was based on the assumptions that recreational users be most likely to be exposed to surficial soils based on their activities. Based on the depth distribution observed for lead, risks from exposure to subsurface soils will be similar or less than those observed for surface soils. However, if concentrations for lead are ever found to increase as a function of depth, the risks based on surface soil exposure will underestimate risks for those individuals exposed to buried materials. The maximum lead concentration in soil/tailings observed at the site at any depth is 21,380 mg/kg.

6.3.2 Uncertainty in Lead Absorption from Soil

Another important source of uncertainty regarding the risk from lead in soil is the degree of absorption (RBA) within the gastrointestinal tract. For this risk assessment, a default relative bioavailability factor for lead of 0.60 has been applied. This introduces uncertainty because the selected value is not based on actual measurements for site soils. Soils are complex by nature and may have numerous attributes which influence overall absorptions characteristics.

6.3.3 Uncertainty in Modeling Approach

All predictive models, including the IEUBK model and the ISE model, are subject to a number of limitations. First, there is inherent difficulty in providing the models with reliable estimates of human exposure to lead-contaminated media. For example, exposure to soil and dust is difficult to quantify because human intake of these media is likely to be highly variable, and it is very difficult to derive accurate measurements of actual intake rates. Second, it is often difficult to obtain reliable estimates of key pharmacokinetic parameters in humans (e.g., absorption fraction, distribution and clearance rates), since direct observations in humans are limited. Finally, the absorption, distribution and clearance of lead in the human body is an extremely complicated process, and any mathematical model intended to simulate the actual processes is likely to be an over-simplification. Consequently, model calculations and predictions are generally rather uncertain.

The Bowers model used to assess lead exposures in youths and adults requires a composite toxicokinetic parameter (the biokinetic slope factor) to predict the effect of exposure on blood lead levels. This value is derived mainly from studies in adult males, and it is not certain that the value is accurate for youths or for women (especially pregnant women). Also, the exposures being modeled with the Bowers model are intermittent rather than continuous, so blood lead levels in the exposed populations are expected to show temporal variability. Toxicity data are not adequate to estimate the level of health risk associated with occasional (rather than continuous) elevations in blood lead level due to intermittent exposures to elevated lead levels in the environment. However, since the observed lead levels in soil/tailings result in predicted blood lead levels that are well below the established level of concern, these uncertainties in the modeling approach do not cast serious doubt on the accuracy of the conclusion that lead levels at this site are not of concern to older children or adults.

7.0 SUMMARY AND CONCLUSIONS

7.1 Risks from Non-Lead COPCs

Interpretation of risk characterization results is a matter of judgement by the risk manager. The measure used to describe the potential for noncarcinogenic toxicity to occur in an individual is expressed by comparing an exposure level over a specified time period with a reference dose derived for a similar exposure period. This ratio of exposure to toxicity is referred to as a hazard quotient. To assess the overall potential for noncarcinogenic effects posed by more than one chemical, these HQs are summed to obtain a hazard index. In general , USEPA considers that acceptable level of excess risk under RME assumptions is an HI equal to or less than one (1E+00) for non-cancer risks. In this case, it is believed that there is no appreciable risk that noncancer health effects will occur. If an HI exceeds 1E+00, there is some possibility that noncancer effects may occur, although an HI above 1E+00 does not indicate an effect will definitely occur. In this instance, it is important to review the contribution of risks from the individual chemicals which were evaluated in the risk assessment.

In evaluating carcinogens, risks are estimated as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to the potential carcinogen. The level of total cancer risk that is of concern is a matter of personal, community and regulatory judgement. In general, it is the policy of the USEPA that remedial action is not warranted where excess cancer risks to the RME individual do not exceed a level of 1E-04 (USEPA, 1991b). It should be noted that, the upper boundary of the risk range is not a discrete line at 1E-04. This risk level may be considered acceptable if justified based on site-specific conditions. However, a risk manager may also decide that a lower level of risk to human health is unacceptable and that remedial action is warranted where, for example, there are uncertainties in the risk assessment results.

A summary of the estimated non-cancer and cancer risks resulting from exposure to arsenic at this site is presented below. reasonable maximum exposure, 0001

Endpoint	Population	Average	RME
Non-Cancer	*Total Risk Low Intensity User*	*1E-02*	*9E-02*
	Total Risk High Intensity User	*6E-03*	*6E-02*
Cancer Risk	*Total Risk Low Intensity User*	*7E-07*	*2E-05*
	Total Risk High Intensity User	*3E-07*	*1E-05*

As seen, none of the non-cancer risks are predicted to exceed a Hazard Index of 1.0. Additionally, no cancer risks are predicted to fall within or below the USEPA's acceptable risk range of 1E-04 and 1E-06. These results indicate that exposure to arsenic is resulting in unacceptable levels of health risk to either low-intensity or high-intensity recreational visitors at this site.

7.2 Risks from Lead

The IEUBK model was utilized to predict the geometric mean blood lead values and P10 values for children exposed either just residential or via a combination of residential and recreational exposure. This approach was used in order to determine the excess blood lead levels attributable to any recreational activities engaged in at this site. The geometric mean blood lead values were predicted to be 1.8 and 2.0 ug/dL for residential and residential plus recreational scenarios, respectively. Although the addition of recreational exposure into the IEUBK model results in higher blood lead levels, the P10 values under this scenario are below USEPA's guideline of 5% and are predicted to range from 0.0% (GSD=1.4) to 0.03% (GSD=1.6), depending on the GSD selected. These results indicate that low-intensity recreational exposures at this site are unlikely to result in blood lead levels in children which result in greater than a 5% probability of exceeding a blood lead level of 10 ug/dL.

The Bowers model was utilized to predict the geometric mean and 95th Percentile blood lead concentrations (PbB_{95}) in visitors who may engage at recreational activities at the site. The predicted geometric mean blood lead values were 1.4 and 1.5 ug/dL, for low intensity and high intensity recreational visitors, respectively. The PbB_{95} concentrations were found to be 4.8 and 5.1 ug/dL for low and high intensity recreational visitors, indicating that recreational activities at the site will not result in blood lead levels with a greater than 5% probability of exceeding a blood lead level of 10 ug/dL.

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TABLES

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Table 3-1
Summary of Analytical Parameters Across Media Types and Sampling Programs

Analytes	Tailings	Soil			Sediment	Groundwater		Surface Water	
		Background	Off-Impoundment	On-Impoundment		Dissolved	Total	Dissolved	Total
Aluminum	2	NONE	NONE	2; 3	1; 2; 3	2; 3; 7	2; 3; 7	1; 2; 5; 6	1; 3; 2; 5; 6
Antimony	2	NONE	NONE	2; 3	1; 2; 3	2; 3; 7	2; 3; 7	1; 2; 6	1; 3; 2; 6
Arsenic	2; 4	2	2	2; 3	1; 2; 3	2; 3; 7	2; 3; 7	1; 2; 5; 6; 7	1; 2; 3; 5; 6; 7
Barium	NONE	2	2	2; 3	3	3; 7	3; 7	5; 6; 7	3; 5; 7
Beryllium	NONE	NONE	NONE	3	3	3; 7	3; 7	NONE	3
Boron	NONE	NONE	NONE	NONE	NONE	NONE	NONE	5	NONE
Cadmium	2; 4	2	2	2; 3	1; 2; 3	2; 3; 7	2; 3; 7	1; 2; 5; 6; 7	1; 2; 3; 5; 6; 7
Calcium	NONE	NONE	NONE	3	3	3; 7	2; 3; 7	5; 6	1; 2; 3; 6
Chromium	2	2	2	2; 3	1; 2; 3	2; 3; 7	2; 3; 7	1; 2; 5; 6; 7	1; 2; 3; 5; 6; 7
Cobalt	NONE	NONE	NONE	3	3	3; 7	3; 7;	NONE	3
Copper	2; 4	2	2	2; 3	1; 2; 3	2; 3; 7	2; 3; 7	1; 2; 5; 6; 7	1; 2; 3; 5; 6; 7
Cyanide	NONE	NONE	NONE	NONE	NONE	NONE	7	NONE	5; 6
Iron	2	NONE	NONE	2; 3	1; 2; 3	2; 3; 7	2; 3; 7	1; 2; 5; 6	1; 2; 3; 5; 6
Lead	2; 4	2	2	2; 3	1; 2; 3	2; 3; 7	2; 3; 7	1; 2; 5; 6; 7	1; 2; 3; 5; 6; 7
Magnesium	NONE	NONE	NONE	3	3	3; 7	2; 3; 7	5; 6	1; 2; 3; 6
Manganese	NONE	NONE	NONE	3	3	2; 3; 7	2; 3; 7	1; 2; 5; 6	1; 2; 3; 5; 6
Mercury	2; 4	2	2	2; 3	1; 2; 3	2; 3; 7	2; 3; 7	1; 2; 5; 6; 7	1; 2; 3; 5; 6; 7
Nickel	NONE	NONE	NONE	3	3	3; 7	3; 7	NONE	3
Phosphorus	NONE	NONE	NONE	NONE	NONE	NONE	2	5	2; 5
Potassium	NONE	NONE	NONE	3	3	3; 7	2; 3; 7	5	1; 2; 3; 6
Selenium	2	2	2	2; 3	1; 2; 3	2; 3; 7	2; 3; 7	1; 2; 5; 6; 7	1; 2; 3; 5; 6; 7
Silver	2; 4	2	2	2; 3	1; 2; 3	2; 3; 7	2; 3; 7	1; 2; 5; 6; 7	1; 2; 3; 5; 6; 7
Sodium	NONE	NONE	NONE	3	3	2; 3; 7	2; 3; 7	5	1; 2; 3; 6
Thallium	NONE	NONE	NONE	3	3	3; 7	3; 7	NONE	3
Vanadium	NONE	NONE	NONE	3	3	3; 7	3; 7	NONE	3
Zinc	2; 4	2	2	2; 3	1; 2; 3	2; 3; 7	2; 3; 7	1; 2; 5; 6; 7	1; 3; 5; 6; 7

<u>Key to Sources</u>
1 = USEPA (2001a) Watershed Study
2 = RMC (2001c) Monthly Monitoring Data
3 = E&E (1993)
4 = USEPA (1991)
5 = STORET
6 = UPCM
7 = RMC (2000a)

Table 3-2: Summary Statistics

Part A: Sediment

Parameter	Detection Frequency	Min* (mg/kg)	Max* (mg/kg)	Avg* (mg/kg)
Aluminum	12/12 (100%)	1,930	28,800	11,844
Antimony	12/12 (100%)	36	99	75
Arsenic	12/12 (100%)	101	310	162
Barium	5/5 (100%)	92	562	276
Beryllium	5/5 (100%)	1.1	2.3	1.8
Cadmium	12/12 (100%)	18	93	52
Calcium	5/5 (100%)	39,800	96,000	58,780
Chromium	12/12 (100%)	15	62	26
Cobalt	5/5 (100%)	5.8	20	14
Copper	12/12 (100%)	173	725	301
Iron	12/12 (100%)	23,000	91,900	39,083
Lead	12/12 (100%)	1,880	6,520	3,453
Magnesium	5/5 (100%)	10,900	14,100	12,960
Manganese	5/5 (100%)	2,200	42,000	10,938
Mercury	12/12 (100%)	0.32	8.2	2.3
Nickel	5/5 (100%)	13	97	45
Potassium	5/5 (100%)	886	4,760	2,847
Selenium	8/12 (67%)	2.5	43	10
Silver	12/12 (100%)	8.0	41	19
Sodium	5/5 (100%)	206.0	1,150	603.4
Thallium	5/5 (100%)	6.6	14	8.6
Vanadium	5/5 (100%)	9.5	71	38
Zinc	12/12 (100%)	2,940	15,200	8,945

Part B: Surface Water

Parameter	Detection Frequency	Min* (mg/L)	Max* (mg/L)	Avg* (mg/L)
Aluminum	57/171 (33%)	0.01	1.4	0.07
Ammonia	34/41 (83%)	0.05	0.97	0.30
Antimony	62/163 (38%)	0.003	0.04	0.005
Arsenic	98/282 (35%)	0.003	0.75	0.008
Barium	108/109 (99%)	0.02	0.22	0.08
Beryllium	5/5 (100%)	0.002	0.002	0.002
Boron	1/1 (100%)	0.06	0.06	0.06
Cadmium	111/278 (40%)	0.001	0.01	0.002
Calcium	166/166 (100%)	39	404	174
Chlorine	90/90 (100%)	44	320	110
Chromium	19/276 (7%)	0.003	0.05	0.007
Chromium, hexavalent	1/1 (100%)	0.001	0.001	0.001
Cobalt	1/5 (20%)	0.003	0.01	0.005
Copper	56/289 (19%)	0.003	0.39	0.008
Cyanide	22/121 (18%)	0.002	0.05	0.003
Fluorides	1/1 (100%)	0.31	0.31	0.31
Iron	130/235 (55%)	0.0002	30	0.31
Lead	250/463 (54%)	0.002	26	0.13
Magnesium	163/163 (100%)	9.1	90	42
Manganese	401/402 (100%)	0.003	12	1.2
Mercury	41/372 (11%)	0.0000001	0.009	0.0005
Nickel	2/5 (40%)	0.01	0.01	0.01
Phosphorus	76/152 (50%)	0.01	0.74	0.05
Potassium	104/153 (68%)	0.25	6.2	2.4
Selenium	23/278 (8%)	0.001	0.02	0.002
Silica	1/1 (100%)	13	13	13
Silver	6/276 (2%)	0.001	0.05	0.003
Sodium	153/153 (100%)	6.7	177	55
Thallium	0/5 (0%)	0.001	0.001	0.001
Vanadium	0/5 (0%)	0.02	0.02	0.02
Zinc	328/330 (99%)	0.01	96	1.2

Part C: Soil and Tailings

Parameter	Detection Frequency	Min* (mg/kg)	Max* (mg/kg)	Avg* (mg/kg)
Arsenic	59/64 (92%)	2.5	243	41
Barium	16/16 (100%)	175	365	241
Cadmium	8/17 (47%)	0.25	96	9.1
Chromium	16/16 (100%)	16	33	22
Copper	18/18 (100%)	13	336	64
Lead	62/62 (100%)	14	5,875	661
Mercury	4/16 (25%)	0.05	3.2	0.32
Selenium	0/16 (0%)	2.5	2.5	2.5
Silver	1/17 (6%)	2.5	22.1	3.7
Zinc	18/18 (100%)	47	14,100	1,378

* Non-Detects evaluated at 1/2 the Detection limit

Table 3-3: Evaluation of Beneficial and Essential Minerals

PART A: EVALUATION OF BENEFICIAL AND ESSENTIAL MINERALS IN SEDIMENT

Chemical	Max Conc[a] mg/kg	TWA-Intake[b] kg/kg-day	Max DI[c] mg/kg-day	RDA[d] mg/kg-day	Ratio DI/RDA	Retain
Calcium	96,000	2.60E-08	2.50E-03	14	<0.001	NO
Chromium III	62	2.60E-08	1.62E-06	1	<0.001	NO
Cobalt	20	2.60E-08	5.20E-07	0.06	<0.001	NO
Copper	725	2.60E-08	1.89E-05	0.037	<0.001	NO
Iron	91,900	2.60E-08	2.39E-03	0.3	0.009	NO
Magnesium	14,100	2.60E-08	3.67E-04	5.7	<0.001	NO
Manganese	42,000	2.60E-08	1.09E-03	0.005	0.218	NO
Potassium	4,760	2.60E-08	1.24E-04	0.57	<0.001	NO
Selenium	43	2.60E-08	1.12E-06	0.005	<0.001	NO
Sodium	1,150	2.60E-08	2.99E-05	34	<0.001	NO
Zinc	15,200	2.60E-08	3.95E-04	0.30	0.001	NO

PART B: EVALUATION OF BENEFICIAL AND ESSENTIAL MINERALS IN SURFACE WATER

Chemical	Max Conc[a] mg/L	TWA-Intake[b] L/kg-day	Max DI[c] mg/kg-day	RDA[d] mg/kg-day	Ratio DI/RDA	Retain
Calcium	404	2.64E-05	1.07E-02	14	<0.001	NO
Chromium III	0.05	2.64E-05	1.32E-06	1	<0.001	NO
Chloride	320	2.64E-05	8.45E-03	0.51	0.017	NO
Cobalt	0.01	2.64E-05	2.75E-07	0.06	<0.001	NO
Copper	0.39	2.64E-05	1.03E-05	0.037	<0.001	NO
Flouride	0.31	2.64E-05	8.18E-06	0.060	<0.001	NO
Iron	30	2.64E-05	7.92E-04	0.3	0.003	NO
Magnesium	90	2.64E-05	2.38E-03	5.7	<0.001	NO
Manganese	12	2.64E-05	3.17E-04	0.005	0.063	NO
Phosphorus	0.74	2.64E-05	1.96E-05	14.000	<0.001	NO
Potassium	6.2	2.64E-05	1.64E-04	0.57	<0.001	NO
Selenium	0.02	2.64E-05	4.49E-07	0.005	<0.001	NO
Sodium	177	2.64E-05	4.67E-03	34	<0.001	NO
Zinc	96	2.64E-05	2.53E-03	0.30	0.008	NO

PART C: EVALUATION OF BENEFICIAL AND ESSENTIAL MINERALS IN SOIL AND TAILINGS

Chemical	Max Conc[a] mg/kg	TWA-Intake[b] kg/kg-day	Max DI[c] mg/kg-day	RDA[d] mg/kg-day	Ratio DI/RDA	Retain
Chromium III	33	5.20E-07	1.72E-05	1	<0.001	NO
Copper	336	5.20E-07	1.75E-04	0.037	0.005	NO
Selenium	2.5	5.20E-07	1.30E-06	0.005	<0.001	NO
Zinc	14,100	5.20E-07	7.33E-03	0.30	0.024	NO

[a] Maximum detected concentration

[b] TWA-Intake = Time-weight average intake rate of environmental medium (RME Low Intensity Recreational Visitor)

Soil: Assumes ingestion of 100 mg/d for 6 years (as 15 kg child) and 500 mg/d for 24 years (as 70 kg adult) for 100 days/yr

Water: Assumes ingestion of 30 mL/hr and dermal contact (3,800 cm^2 skin surface area for child and 5,000 cm^2 for adult) and 1.5 hours/day for 6 years (as 15 kg child) and 2 L/d for 24 years (as 70 kg adult) for 10 days/yr

[c] DI = Daily intake of chemical (mg/kg-day)

[d] RDA = Recommended Dietary Allowance or Toxicity Value from USEPA (1994a)

Sodium value based on 2,400 mg/day recommended daily allowance divided by 70 kg body weight

Table 3-4: Comparison of Detection Limits to Risk Based Concentrations

Part A: Sediment

Parameter	Detection Frequency	Non-Detect Range (ppm)	RBC (ppm)	DL Adequate?	Retain?
Aluminum	12/12 (100%)	--	7,800	YES	**YES**
Antimony	12/12 (100%)	--	3.1	YES	**YES**
Arsenic	12/12 (100%)	--	0.04	YES	**YES**
Barium	5/5 (100%)	--	550	YES	**YES**
Beryllium	5/5 (100%)	--	16	YES	**YES**
Cadmium	12/12 (100%)	--	7.8	YES	**YES**
Lead	12/12 (100%)	--	400	YES	**YES**
Mercury	12/12 (100%)	--	2.2	YES	**YES**
Nickel	5/5 (100%)	--	160	YES	**YES**
Silver	12/12 (100%)	--	39	YES	**YES**
Thallium	5/5 (100%)	--	0.55	YES	**YES**
Vanadium	5/5 (100%)	--	55	YES	**YES**

Part B: Surface Water

Parameter	Detection Frequency	Non-Detect Range (ppm)	RBC (ppm)	DL Adequate?	Retain?
Aluminum	57/171 (33%)	0.0171 - 0.05	3,700	YES	YES
Ammonia	34/41 (83%)	0.1	21	YES	YES
Antimony	62/163 (38%)	0.005 - 0.0243	1.5	YES	YES
Arsenic	98/282 (35%)	0.005 - 0.02	0.45	YES	YES
Barium	108/109 (99%)	0.1	260	YES	YES
Beryllium	5/5 (100%)	–	7.3	YES	YES
Boron	1/1 (100%)	–	329	YES	YES
Cadmium	111/278 (40%)	0.001 - 0.005	1.8	YES	YES
Chlorine	90/90 (100%)	–	0.04	YES	YES
Chromium VI	1/1 (100%)	–	11	YES	YES
Cyanide	22/121 (18%)	0.004 - 0.008	73	YES	YES
Lead	250/463 (54%)	0.003 - 0.1	4.0	YES	YES
Mercury	41/372 (11%)	0.0000002 - 0.005	1.1	YES	YES
Nickel	2/5 (40%)	0.0111	*73	YES	YES
Silica	1/1 (100%)	–		YES	YES
Silver	6/276 (2%)	0.002 - 0.1	18	YES	NO
Thallium	0/5 (0%)	0.0016	0.26	YES	NO
Vanadium	0/5 (0%)	0.0357	26	YES	NO

* Based on Region 9 PRG value for tap water

Part C: Soil and Tailings

Parameter	Detection Frequency	Non-Detect Range (ppm)	RBC (ppm)	DL Adequate?	Retain?
Arsenic	59/64 (92%)	5	0.04	YES	YES
Barium	16/16 (100%)	–	550	YES	YES
Cadmium	8/17 (47%)	0.5	7.8	YES	YES
Chromium III	16/16 (100%)	–	23	YES	YES
Copper	16/16 (100%)	–	310	YES	YES
Lead	62/62 (100%)	–	400	YES	YES
Mercury	4/16 (25%)	0.1	2.2	YES	YES
Silver	1/17 (6%)	5	39	YES	YES

Table 3-5: Maximum and Average Chemical Concentrations in Soil and Background Concentrations in the United States

Chemical	Max Soil Conc (mg/kg)	Avg Soil Conc (mg/kg)	Background Concentrations for Soils in the Western United States*		Background Concentrations for Soils in the United States**	Background Concentrations for Soils in the United States***		Retain?
			Range (ppm)	Geometric Mean	Range (ppm)	Range (ppm)	Mean	
Arsenic	243	41	<0.10 - 97	5.5	1.0 - 40	1 - 40	5.0	YES
Barium	365	241	70 - 5,000	580	100 - 3,500	15 - 3,000		NO
Cadmium	96	9.1	<150 - 300	65	0.01 - 7.0		0.25	YES
Lead	5,875	661	<10 - 700	17	2.0 - 200			YES
Mercury	3.2	0.32	<0.01 - 4.6	0.05	0.01 - 0.08	0.02 - 0.625		YES
Silver	22.1	3.7			0.1 - 5.0			YES

* Based on Shacklette and Boerngen, 1984
** Based on Dragun, 1988
*** Based on ATSDR, 1997

Table 3-6: Maximum Chemical Concentrations and Risk-Based Concentrations for Recreational Users

Part A: Sediment

Chemical	Max Sediment Conc (mg/kg)	Calculated RBC* (mg/kg)	Retain as COPC?
Aluminum	28,800	3,832,463	NO
Antimony	99	1,533	NO
Arsenic	310	75	**YES**
Barium	562	268,275	NO
Beryllium	2.3	7,665	NO
Cadmium	93	3,832	NO
Lead	6,520	400	**YES**
Manganese	42,000	536,550	NO
Mercury	8.2	1,150	NO
Nickel	97	76,650	NO
Silver	41	19,163	NO
Thallium	13.6	307	NO
Vanadium	71	34,493	NO

Part B: Surface Water

Chemical	Max Surface Water Conc (mg/L)	Calculated RBC* (mg/L)	Retain as COPC?
Aluminum	1.4	3,788	NO
Ammonia	0.97	209*	NO
Antimony	0.04	2	NO
Arsenic	0.8	0.07	**YES**
Barium	0.22	265	NO
Beryllium	0.002	8	NO
Boron	0.06	341	NO
Cadmium	0.01	4	NO
Chlorine	320	379	NO
Chromium VI	0.001	11	NO
Cyanide	0.05	76	NO
Lead	26	4.0	**YES**
Mercury	0.009	1	NO
Nickel	0.006	76	NO

Part C: Soil and Tailings

Chemical	Max Soil/Tailing Conc (mg/kg)	Calculated RBC* (mg/kg)		Minimum Calculated RBC	Retain as COPC?
		RME low-intensity visitor	RME high-intensity visitor		
Arsenic	243	4	6	4	**YES**
Cadmium	96	192	255	192	NO
Lead	5,875	400		400	**YES**
Mercury	3	57	77	57	NO
Silver	22	40,379,305	53,839,601	40,379,305	NO

* Based on HQ = 0.1 or Risk = 1E-06

* Based on Region 3 RBC

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FIGURES

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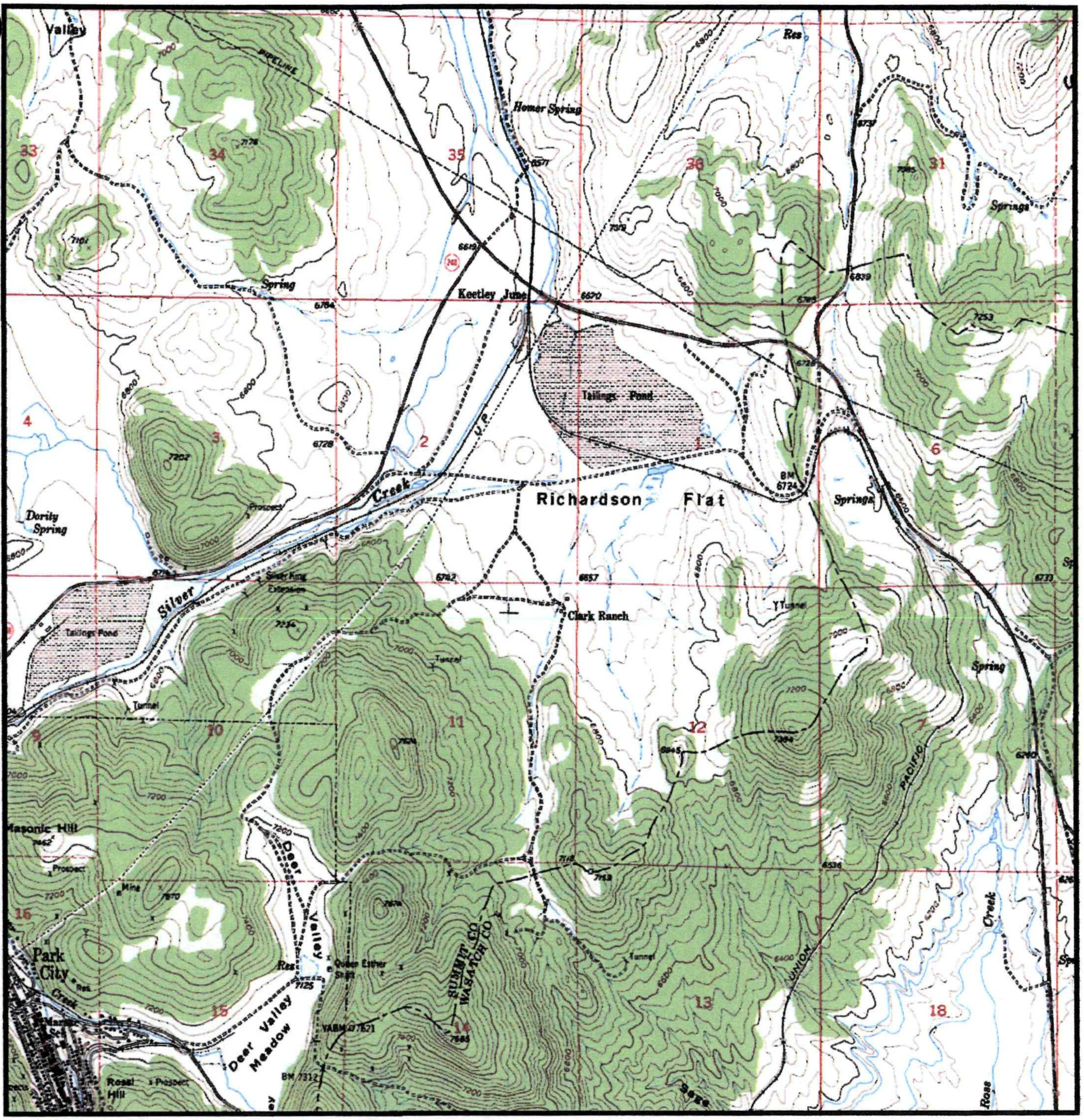


Figure 1 - 1
Richardson Flat Tailings Site Location Map

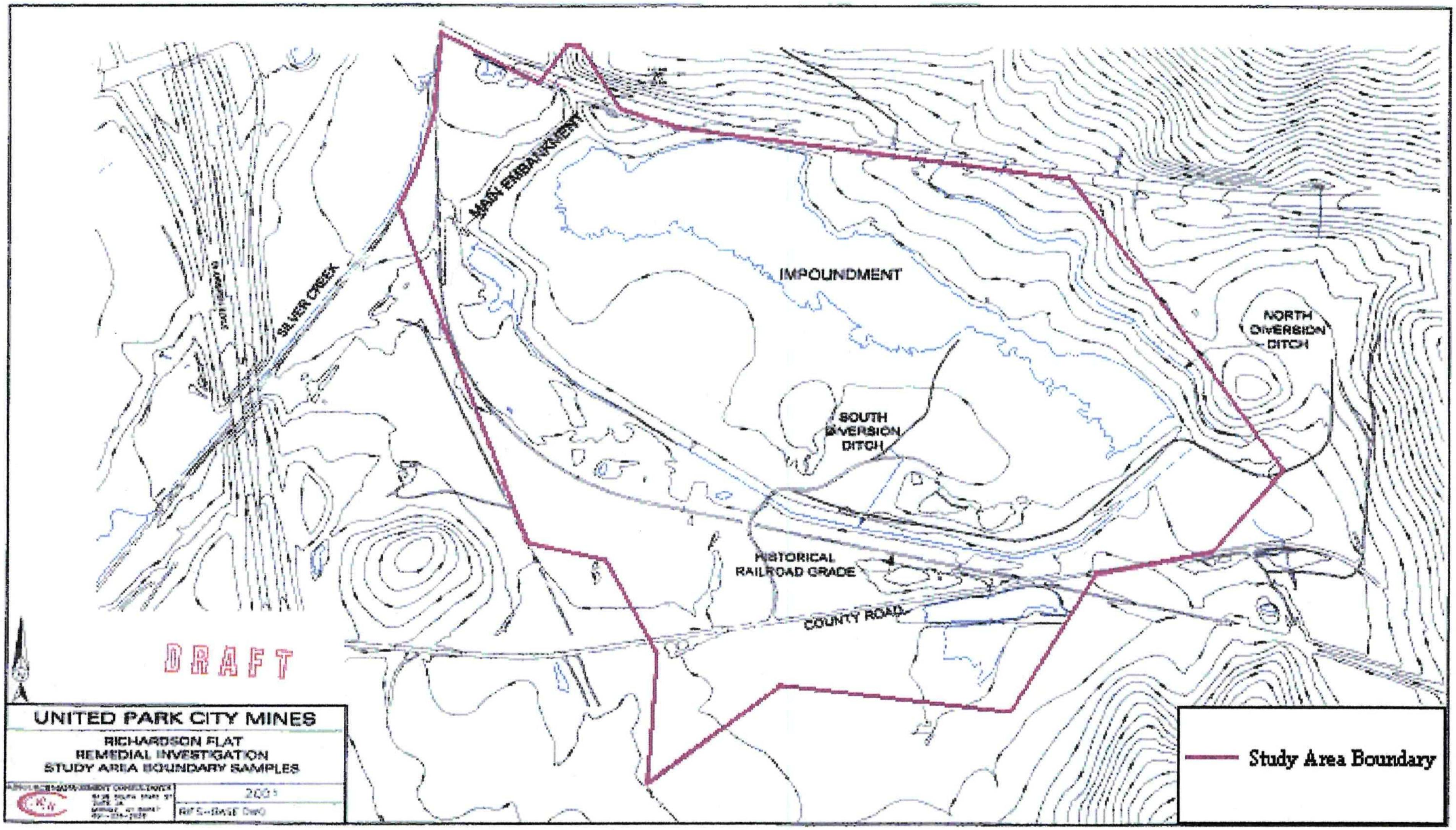
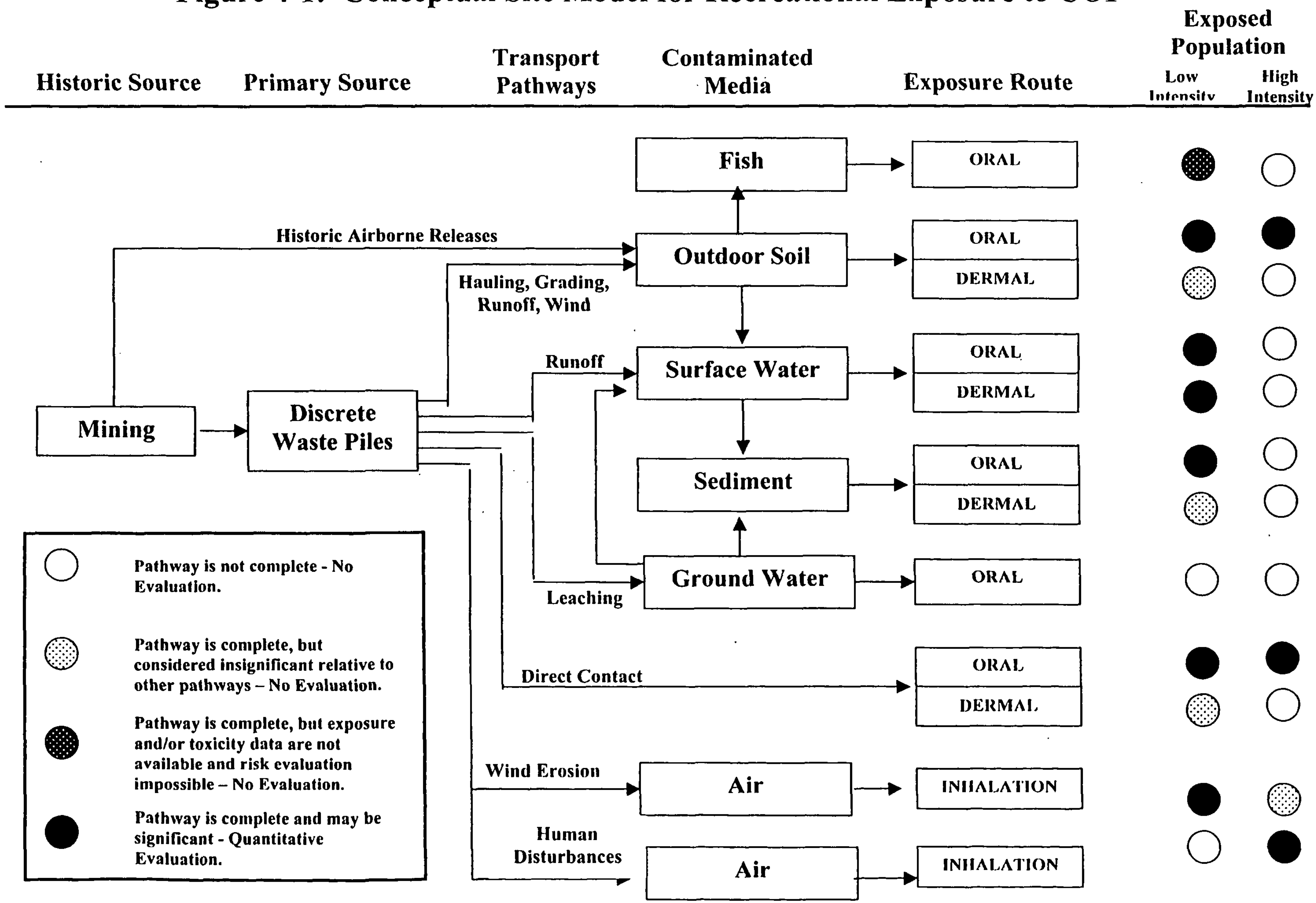


Figure 3-1
Richardson Flats Tailings Study Area Boundary

Source: RMC, 2001 [DRAFT RI/FS]

Figure 4-1: Conceptual Site Model for Recreational Exposure to COPCs



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APPENDIX A

RAW DATA SUMMARY

electronic data will be provided upon request

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APPENDIX B

EXPOSURE ASSUMPTIONS

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EXPOSURE ASSUMPTIONS
RICHARDSON FLATS TAILING SITE

March 2003

Prepared for the:

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1.0 SITE DESCRIPTION

The Richardson Flats Tailing (RFT) Site is located 1.5 miles northeast of Park City, Utah occupying about 700 acres in a small valley in Summit County, Utah. The RFT site is part of the Park City Mining District where silver-laden ore was mined and milled from the Keetley Ontario Mine as well as other mining operations. Tailings were deposited into an impoundment covering 160 acres of the 700 acre property just east of Silver Creek. Tailings were deposited to the impoundment from the mill by use of a slurry pipeline from 1975 through 1981. Mining and milling operations ended in 1982.

2.0 LAND USE

The site is located in a rural area whose topography is characterized by a broad valley with undeveloped rangeland. Silver Creek is located within a few hundred feet from the main tailings impoundment. Typical land use is limited to recreational purposes. It is not envisioned, for the purposes of the human health risk assessment, that this property will be developed for residential purposes. However, it is envisioned that modifications to the site as a recreational park could be implemented.

There are a wide variety of different recreational activities which people may engage in at this site, and hence there are a wide variety of different recreational exposure scenarios which might warrant evaluation. Two separate scenarios were considered to serve as the representative population evaluated:

- low intensity uses such as, hiking, biking, and picnicking
- high intensity uses such as, horseback riding, dirt-biking, soccer and baseball

3.0 EXPOSURE SCENARIOS

3.1 Recreational Visitor – Low Intensitv Activities

This scenario envisions an open-space visitor who engages in lower intensity activities at the site, including; hiking, biking, and picnicking. Potential pathways of exposure include:

- ingestion of tailings/soil
- inhalation of particulates

It is assumed that this low intensity recreational visitor may occasionally be exposed to surface water and sediments at or near the site. These pathways are further discussed in Section 3.3.

3.2 Recreational Visitor – High Intensity Activities

This scenario envisions a recreational site visitor who engages in higher intensity activities at the site, including; horseback riding, soccer, baseball. Potential pathways of exposure include:

- ingestion of tailings/soil
- inhalation of particulates

3.3 Exposure to Surface Water & Sediment

Exposure of low intensity recreational visitors to surface water and sediment at the site are being evaluated separately at the request of the site RPM. Two locations where exposure might occur to surface water and sediment include: onsite ponded water areas and Silver Creek. Each of these locations will be evaluated separately for the recreational user who may frequent these water sources on occasion. Potential pathways of exposure include:

- ingestion of sediment
- dermal contact with water
- ingestion of surface water

Recreational visitors can get contaminated soil/tailings/sediments on their skin while engaging in recreational activities. Dermal contact with contaminated soil is of potential health concern mainly because some chemicals can be absorbed across the skin into the blood, but dermal irritation (e.g., due to contact with acidic tailings) may also occur. Even though information is limited on the rate and extent of dermal absorption of metals in soil across the skin, most scientists consider that this pathway is likely to be minor in comparison to the amount of exposure that occurs by soil and dust ingestion. This view is based on the following concepts: 1) most people do not have extensive and frequent direct contact with soil, 2) most metals tend to bind to soils, reducing the likelihood that they would dissociate from the soil and cross the skin, and 3) ionic species such as metals have a relatively low tendency to cross the skin even when contact does occur. These presumptions are supported by screening level calculations which indicate that dermal exposure of most metals is likely to be no larger (and probably much lower) than absorption due to soil ingestion. Based on these considerations, along with a lack of data to allow reliable estimation of dermal uptake of metals from soil, USEPA Region 8 generally recommends that dermal exposure to metals in soils not be evaluated quantitatively (USEPA, 1995). Therefore, this pathway will not be evaluated quantitatively in the risk assessment.

4.0 EXPOSURE ASSUMPTIONS FOR NON-LEAD COPCS

The following pages provide draft exposure parameters for each of the populations and each of the scenarios outlined above. Whenever possible the draft value is based on standard default EPA guidance. Some values, however, remain based on professional judgment or reflect those used at similar sites. All of these parameters should be reviewed and subjected to a site-specific reality check. Input and suggestions from all concerned parties is requested.

For every exposure pathway of potential concern, it is expected that there will be differences between different individuals in the level of exposure at a specific location due to differences in intake rates, body weights, exposure frequencies, and exposure durations. Thus, there is normally a wide range of average daily intakes between different members of an exposed population. Because of this, all daily intake calculations must specify what part of the range of doses is being estimated. Typically, attention is focused on intakes that are "average" or are otherwise near the central portion of the range, and on intakes that are near the upper end of the range (e.g., the 95th percentile). These two exposure estimates are referred to as Central Tendency Exposure (CTE) and Reasonable Maximum Exposure (RME), respectively.

The USEPA has collected a wide variety of data and has performed a number of studies to help establish default values for most residential and worker exposure parameters. The chief sources of these standard default values are the following documents:

1. Risk Assessment Guidance for Superfund (RAGS). Volume I. Human Health Evaluation Manual (Part A). EPA 1989.

2. Human Health Evaluation Manual, Supplemental Guidance: "Standard Default Exposure Factors". EPA 1991.

3. Superfund's Standard Default Exposure Factors for the Central Tendency and Reasonable Maximum Exposure. Draft. EPA 1993.

4. Exposure Factors Handbook. Update to Exposure Factors Handbook EPA. 1997.

The following sections list the exposure parameters recommended for evaluation of low and high intensity recreational visitors by inhalation, ingestion of and dermal contact with surface water, and incidental ingestion of soil or sediment, along with the resulting HIF terms for CTE and RME exposureDue to the lack of site specific data on the frequency of recreational use of the Richardson Flat Tailings Site, an open space usage survey in Jefferson County, Colorado (Jefferson County Open Space Department, 1996) were used to estimate the exposure frequency (EF) for recreational visitors at the Richardson Flats Tailings Site. During 1996, 779 individuals were interviewed and asked to quantify the number of times per year they visited Open Space Parks in Jefferson County. The arithmetic mean (39 visits/year) and 90th percentile (100 visits/year) of the total number of visits per year were calculated from the survey results and are used as the CTE and RME exposure frequency assumptions, respectively, for the Richardson Flats Site. The CTE and RME exposure frequencies were multiplied by an additional parameter, fraction of exposure at the site (FS), to adjust for the potential use of additional open spaces, other than the Richardson Flats Site, for recreation. In the absence of any site-specific data, the CTE and RME values for the FS parameter were set to 0.5 and 1.0, respectively, based on professional judgement. Theses values are thought to be appropriate for both CTE and RME scenarios by assuming that 50% and 100% of all recreational visits, respectively, occur at the Richardson Flats Tailings Site. Thus, 19.5 visits/year (CTE) and 100 visits per year (RME) are used as the exposure frequency assumptions at the site.

4.1 Recreational Visitor – Low Intensitv Activities

Receptor Population:	combined child (1 - 6 yrs) and adult (7+ yrs)
Exposure Frequency:	19.5 days/year (CTE), 100 days/year (RME), (Jefferson County Open Space Department, 1996 and Professional Judgment)
Health Endpoint:	cancer (chronic exposure), non-cancer
Exposure Pathways:	soil/tailing ingestion, inhalation of particulates

4.1.1 Soil/Tailings Ingestion

Both chronic and lifetime average intake rates are time-weighted to account for the possibility that an adult may begin exposure as a child (EPA 1989, 1991, 1993), as follows:

$$TWA - DIs = Cs\left(\frac{IRc}{BWc}\bullet\frac{EFc\bullet EDc}{(ATc+ATa)}+\frac{IRa}{BWa}\bullet\frac{EFa\bullet EDa}{(ATc+ATa)}\right)$$

where:

$TWA\text{-}DI_s$ = Time-weighted Daily Intake from ingestion of soil/tailings (mg/kg-d)
C_s = Concentration of chemical in soil/tailings (mg/kg)
IR = Intake rate (kg/day) when a child (IR_c) or an adult (IR_a)
BW = Body weight (kg) when a child (BW_c) or an adult (BW_a)
EF = Exposure frequency (days/yr) when a child (EF_c) or an adult (EF_a)
ED = Exposure duration (years) when a child (ED_c) or an adult (ED_a)
AT = Averaging time (days) while a child (AT_c) or an adult (AT_a)

Default values and assumptions recommended by EPA (1989, 1991, 1993) for evaluation of exposure to soil/tailings are listed below. There are no data on ingestion rates of tailings by children or adults while engaged in recreational activities at this site. Therefore, based on professional judgment, ingestion rates of soil/tailings of 50 mg/day and 100 mg/day are assumed for adult and child RME low intensity visitors respectively. For CTE visitors, these values were assumed to be half of that attributable to the RME exposure (25 mg/day and 50 mg/day). Assuming an approximate site visit of 2 hours, these values (RME: 25 mg/hr child, 50 mg/hr adult) are approximately equal to 4 times the levels of soil that a resident is expected to ingest on an hourly basis. The RME default ingestion value for a residential child is 200 mg/day and 100 mg/day for an adult, based on a 16 hour day. This is equivalent to 12.5 mg/hr for a resident child and 6.3 mg/hr for a resident adult. Since it is expected that a recreational visitor will consume more soil than a typical resident on an hourly basis, these values are judged appropriate for use at this site.

Exposure Parameters for Soil/Tailings Ingestion	CTE		RME	
	Child	Adult	Child	Adult
IR (kg/event)	50	25	100	50
BW (kg)	15	70	15	70
EF (events/year)	19.5	19.5	100	100
ED (years)	2	7	6	24
AT (non-cancer effects) (days)	2*365	7*365	6*365	24*365
AT (cancer effects) (days)	--	70*365	--	70*365

Based on the exposure parameters above, the HIFs for exposure of children and adults to soil/tailings are as follows:

Recreational Exposure to Soil/Tailings	HIF (kg/kg-d)	
	CTE	RME
TWA-chronic (non-cancer)	5.4E-08	5.2E-07
TWA-lifetime (cancer)	7.0E-09	2.2E-07

4.1.2 Inhalation of Particulates

The basic equation recommended by EPA (1989) for evaluation of risks due to inhalation exposure to a chemical in air is:

$$TWA - DIa = Ca\left(\frac{IRc}{BWc} \bullet \frac{ETc \bullet EFc \bullet EDc}{(ATc + ATa)} + \frac{IRa}{BWa} \bullet \frac{ETa \bullet EFa \bullet EDa}{(ATc + ATa)}\right)$$

where:

$TWA\text{-}DI_a$	=	Time-weighted Daily Intake from inhalation of a chemical in air (mg/kg-day)
C_a	=	Concentration of chemical in air (mg/m^3)
IR	=	Breathing rate of air (m^3/hour) when a child (IR_c) or an adult (IR_a)
ET	=	Exposure time (hours/day) when a child (ET_c) or an adult (ET_a)
EF	=	Exposure frequency (days/yr) when a child (EF_c) or an adult (EF_a)
ED	=	Exposure duration (years) when a child (ED_c) or an adult (ED_a)
AT	=	Averaging time (days) while a child (AT_c) or an adult (AT_a)
BW	=	Body weight (kg) when a child (BW_c) or an adult (BW_a)
AT	=	Averaging time (days)

Default values and assumptions recommended by EPA (1989, 1991, 1993) for evaluation of exposure to particulates in air are listed below. Inhalation rates of 1.6 m^3/hr for children and 2.4 m^3/hr for adults are based on the average of medium and heavy activity inhalation rates for these age groups. This information is from the 1997 Exposure Factors Handbook and was used as inputs in the Rocky Flats Task 3 Report (EPA, 2001a). The Exposure Time was based on the 1995 Boulder County open space survey (Boulder County Open Space Operations, 1995) of time spent on site (19% < 1 hour, 71% 1-3 hours, 9% 4-6 hours, and 1% >7 hours). Values of 1.5 and 2.5 hours/day were selected for the CTE and RME exposures, respectively. Although this information pertains to a different site, the values are judged to be applicable at Richardson Flats. The exposure frequency is estimated to be 19.5 days per year for CTE individuals and 100 days per year for RME individuals, based on the mean (39 visits per year for CTE) and 90th percentile (100 visits per year for RME) of visits to Jefferson County Open Space (Jefferson County Open Space Department, 1996) and the assumption that 50% (CTE) and 100% (RME) of all visits occur at the Richardson Flats site.

Exposure Parameters for Inhalation of Particulates	**CTE**		**RME**	
	Child	**Adult**	**Child**	**Adult**
IR (m^3/hr)	1.6	2.4	1.6	2.4
BW (kg)	15	70	15	70
ET (hr/day)	1.5	1.5	2.5	2.5
EF (days/yr)	19.5	19.5	100	100
ED (years)	2	7	6	24
AT (non-cancer effects) (days)	2*365	7*365	6*365	24*365
AT (cancer effects) (days)	--	70*365	--	70*365

Based on the exposure parameters above, the HIFs for exposure of children and adults to particulates are as follows:

Recreational Exposure to Particulates	HIF (m^3/kg-d)	
	CTE	RME
TWA-chronic (non-cancer)	4.0E-03	3.3E-02
TWA-lifetime (cancer)	5.2E-04	1.4E-02

4.2 Recreational Visitor – High Intensity Activities

Receptor Population: Adult (7+ yrs)
Exposure Frequency: 19.5 days/year (CTE), 100 days/year (RME), (Jefferson County Department of Open Space, 1996 and Professional Judgment)
Health Endpoint: cancer (chronic exposure), non-cancer
Exposure Pathways: soil/tailing ingestion, inhalation of particulates

4.2.1 Soil/Tailings Ingestion

The basic equation used to assess risks from incidental ingestion of tailings or contaminated soil by recreational visitors is as follows:

$$DIs = Cs\left(\frac{IR}{BW}\right)\left(\frac{EF \bullet ED}{AT}\right)$$

where:

DI_s	=	Daily intake of chemical from ingestion of soil/tailings (mg/kg-d)
C_s	=	Concentration of chemical in soil/tailings (mg/kg)
IR_t	=	Intake rate (kg/day)
BW	=	Body weight of the exposed person (kg)
EF	=	Exposure frequency (days/year)
ED	=	Exposure duration (years)
AT	=	Averaging time (days)

There are no data on ingestion rates of tailings by adults while engaged in high intensity recreational activities at this site. Therefore, based on professional judgment, ingestion rates of soil/tailings of 50 mg/day and 100 mg/day are assumed for CTE and RME exposure, respectively. Assuming an approximate site visit of 2 hours, these values (25 mg/hr CTE, 50 mg/hr RME) are approximately equal to 8 times the levels of soil that an adult resident is expected to ingest on an hourly basis. The RME default ingestion value for a residential adult is 100 mg/day for an adult, or 6.3 mg/hr based on a 16 hour day. Since it is expected that a recreational visitor will consume more soil than a typical resident on an hourly basis, these values are judged appropriate for use at this site. Additionally, since it is expected that higher intensity activities will lead to increased ingestion of soil/tailings, these values are 2-fold higher than those selected for use under the low-intensity activity scenario. The exposure frequency is estimated to be 19.5 days per year for CTE individuals and 100 days per year for RME individuals, based on the mean (39 visits per year for CTE) and 90th percentile (100 visits per year for RME) of visits to Jefferson County Open Space (Jefferson County Open Space Department, 1996) and the assumption that 50% (CTE) and 100% (RME) of all visits occur at the Richardson Flats site.

The exposure parameters are summarized below:

Exposure Parameter for Soil/Tailings Ingestion	CTE	RME
IR (kg/event)	50	100
BW (kg)	70	70
EF (events/year)	19.5	100
ED (years)	7	24
AT (non-cancer effects) (days)	7·365	24·365
AT (cancer effects) (days)	70·365	70·365

Based on these exposure parameters, the HIF values for exposure of high intensity recreational visitors to tailings and contaminated soil are as follows:

Recreational Exposure to Soil/Tailings	HIF (kg/kg-d)	
	CTE	RME
Chronic (non-cancer)	3.8E-08	3.9E-07
Lifetime (cancer)	3.8E-09	1.3E-07

4.2.2 Inhalation of Particulates

The basic equation recommended by EPA (1989) for evaluation of risks due to inhalation exposure to a chemical in air is:

$$DIa = Ca \bullet \left(\frac{BR}{BW}\right) \bullet \left(\frac{ET \bullet EF \bullet ED}{AT}\right)$$

where:

DI_a	=	Daily Intake from inhalation of a chemical in air (mg/kg-d)
C_a	=	Concentration of chemical in air (mg/m^3)
BR	=	Breathing rate of air (m^3/hour)
ET	=	Exposure time (hours/day)
EF	=	Exposure frequency (days/year)
ED	=	Exposure duration (years)
BW	=	Body weight (kg)
AT	=	Averaging time (days)

Default values and assumptions recommended by EPA (1989, 1991, 1993) for evaluation of exposure to particulates in air are listed below. An inhalation rate of 2.4 m^3/hr for adults was based on the average of medium and heavy activity inhalation rates for this age group. This information is from the 1997 Exposure Factors Handbook and was used as inputs in the Rocky Flats Task 3 Report (EPA, 2001a). The Exposure Time was based on the 1995 Boulder County open space survey (Boulder County Open Space Operations, 1995) of time spent on site (19% < 1 hour, 71% 1-3 hours, 9% 4-6 hours, and 1% >7 hours). Values of 1.5 and 2.5 hours/day were selected for the CTE and RME exposures, respectively. Although this information pertains to a different site, the values are judged to be applicable at Richardson Flats. The exposure frequency is estimated to be 19.5 days per year for CTE individuals and 100 days per year for RME individuals, based on the mean (39 visits per year for CTE) and 90th percentile (100 visits per year for RME) of visits to Jefferson County Open Space (Jefferson County Open Space Department, 1996) and the assumption that 50% (CTE) and 100% (RME) of all visits occur at the Richardson Flats site.

Exposure Parameters for Inhalation of Particulates	**CTE**	**RME**
BR (m^3/hr)	2.4	2.4
BW (kg)	70	70
ET (hr/day)	1.5	2.5
EF (days/yr)	19.5	100
ED (years)	7	24
AT (non-cancer effects) (days)	7·365	24·365
AT (cancer effects) (days)	70·365	70·365

Based on the exposure parameters above, the HIFs for exposure to particulates are as follows:

Recreational Exposure to Particulates	HIF (m^3/kg-d)	
	CTE	RME
Chronic (non-cancer)	2.7E-03	2.3E-02
Lifetime (cancer)	2.7E-04	8.1E-03

4.3 Exposure to Surface Water & Sediment

Receptor Population: combined child (1 - 6 yrs) and adult (7+ yrs)
Exposure Frequency: 2 days/year (CTE), 10 days/year (RME): this assumes that the low intensity visitor is exposed to these media during 1 out of every 10 standard site visits, with 50% and (CTE) and 100% of all visits occurring at the Richardson Flats Site
Health Endpoint: cancer (chronic exposure), non-cancer
Exposure Pathways: ingestion of sediments, dermal contact with surface water, ingestion of surface water

4.3.1 Ingestion of Sediments

The basic equation used to assess risks from incidental ingestion of sediments by recreational visitors while visiting water areas is as follows. Both chronic and lifetime average intake rates are time-weighted to account for the possibility that an adult may begin exposure as a child (EPA 1989, 1991, 1993):

$$TWA - DIs = Cs\left(\frac{IRc}{BWc} \bullet \frac{EFc \bullet EDc}{(ATc + ATa)} + \frac{IRa}{BWa} \bullet \frac{EFa \bullet EDa}{(ATc + ATa)}\right)$$

where:

$TWA\text{-}DI_s$ = Time-weighted Daily Intake from ingestion of sediment (mg/kg-d)
C_s = Concentration of chemical in sediment (mg/kg)
IR = Intake rate (kg/day) when a child (IR_c) or an adult (IR_a)
BW = Body weight (kg) when a child (BW_c) or an adult (BW_a)
EF = Exposure frequency (days/yr) when a child (EF_c) or an adult (EF_a)
ED = Exposure duration (years) when a child (ED_c) or an adult (ED_a)
AT = Averaging time (days) while a child (AT_c) or an adult (AT_a)

There are no data on ingestion rates of sediments by visitors while engaged in recreational activities along the river or in ponded water areas at the site. Therefore, in the absence of data, ingestion rates of soil/tailings of 25 mg/day and 50 mg/day are assumed for adult and child RME visitors respectively. For CTE visitors, these values were assumed to be half of that attributable to the RME exposure (12.5 mg/day and 25 mg/day). This is equivalent to half of the quantity consumed by the low intensity recreational visitor from soil/tailings ingestion. The exposure frequency is estimated to be 2 days per year for CTE individuals and 10 days per year for RME individuals, based on the assumption that the low intensity visitor is exposed to these media during 1 out of every 10 standard visits (4 visits per year (CTE) and 10 visits per year (RME)) and that 50% (CTE) and 100% (RME) of all visits occur at the Richardson Flats site. The exposure parameters are summarized below:

Exposure Parameters for Ingestion of Sediments	**CTE**		**RME**	
	Child	**Adult**	**Child**	**Adult**
IR (kg/day)	25	12.5	50	25
BW (kg)	15	70	15	70
EF (days/year)	2	2	10	10
ED (years)	2	7	6	24
AT (non-cancer effects) (days)	2*365	7*365	6*365	24*365
AT (cancer effects) (days)	--	70*365	--	70*365

Based on these exposure parameters, the HIF values for exposure of visitors to sediments are as follows:

Recreational Exposure to Sediments	**HIF (kg/kg-d)**	
	Average	**RME**
Chronic (non-cancer)	2.8E-09	2.6E-08
Lifetime (cancer)	3.6E-10	1.1E-08

4.3.2 Dermal Contact with Surface Water

The basic equation recommended by EPA (1989) for evaluation of dermal exposure to a chemical dissolved in water is as follows. Both chronic and lifetime average intake rates are time-weighted to account for the possibility that an adult may begin exposure as a child (EPA 1989, 1991, 1993):

$$ADsw = Csw\left(\frac{SAc \bullet PC \bullet ETc \bullet 1E-03}{BWc} \bullet \frac{EFc \bullet EDc}{(ATc + ATa)} + \frac{SAa \bullet PC \bullet ETa \bullet 1E-03}{BWa} \bullet \frac{EFa \bullet EDa}{(ATc + ATa)}\right)$$

where:

AD_{sw}	=	Absorbed dose from dermal contact with surface water (mg/kg-d)
C_{sw}	=	Concentration of chemical in surface water (mg/L)
SA	=	Surface area exposed (cm^2) for child (SA_c) or adult (SA_a)
PC	=	Chemical-specific permeability constant (cm/hr)
ET	=	Exposure time (hr/day) for child (ET_c) or adult (ET_a)
1E-03	=	Conversion factor (L/cm^3)
EF	=	Exposure frequency (days/yr) child (EF_c) or adult (EF_a)
ED	=	Exposure duration (yrs) for child (ED_c) or adult (ED_a)
BW	=	Body weight (kg) child (BW_c) or adult (BW_a)
AT	=	Averaging time (days) for child (AT_c) or adult (AT_a)

It is assumed that dermal exposure of a recreation visitor to water occurs mainly while wading near the river edge or ponded areas, and that dermal contact is mainly restricted to the lower extremities (upper and lower legs and feet) as well as the hands. The surface area for these body parts in children and adults is the 50th percentile for hands, arms, and lower legs (EPA, 1997) (SAF, 2000). No site-specific data on recreation frequency or duration of wading activities per trip are available, so values of 2 (CTE) to 10 (RME) days/year, and 0.5 (CTE) to 1.5 (RME) hours/day are assumed. The exposure time is based on the FE Warren site (SAF, 2000), where estimated time spent in surface waters were evaluated. The exposure frequency is based on the assumption that the low intensity visitor is exposed to these media during 1 out of every 10 standard visits (4 visits per year (CTE) and 10 visits per year (RME)) and that 50% (CTE) and 100% (RME) of all visits occur at the Richardson Flats site. The value of PC is chemical specific, and few measured values are available for metals. Therefore, the EPA (1992b) suggests using a PC value of 1E-03 cm/hr as a conservative estimate. Other exposure parameters are the same as described above. These exposure parameters are summarized below.

Exposure Parameters for Dermal Contact with Surface Water	CTE		RME	
	Child	Adult	Child	Adult
SA (cm^2)	3,800	5,000	3,800	5,000
PC (cm/hr)	1E-03	1E-03	1E-03	1E-03
BW (kg)	15	70	15	70
ET (hours/day)	0.5	0.5	1.5	1.5
EF (days/year)	2	2	10	10
ED (years)	2	7	6	24
AT (non-cancer effects) (days)	2*365	7*365	6*365	24*365
AT (cancer effects) (days)	--	70*365	--	70*365

Based on these exposure parameters, the HIF values for dermal exposure of low intensity recreational visitors to surface water are as follows:

Recreational Exposure for Dermal Contact with Surface Water	HIF (kg/kg-d)	
	Average	RME
Chronic (non-cancer)	3.1E-07	4.4E-06
Lifetime (cancer)	3.9E-08	1.9E-06

4.3.3 Ingestion of Surface Water

The basic equation for evaluation of exposure from ingestion of surface water while participating in water-based recreational activities is as follows. Both chronic and lifetime average intake rates are time-weighted to account for the possibility that an adult may begin exposure as a child (EPA 1989, 1991, 1993):

$$TWA - DIw = Cw\left(\frac{IRc}{BWc}\bullet\frac{ETc\bullet EFc\bullet EDc}{(ATc+ATa)}+\frac{IRa}{BWa}\bullet\frac{ETa\bullet EFa\bullet EDa}{(ATc+ATa)}\right)$$

where:

$TWA\text{-}DI_s$ = Time-weighted Daily Intake from ingestion of water (mg/kg-d)
C_s = Concentration of chemical in surface water (mg/L)
IR = Intake rate (L/day) when a child (IR_c) or an adult (IR_a)
BW = Body weight (kg) when a child (BW_c) or an adult (BW_a)
ET = Exposure time (hours/day) when a child (ET_c) or an adult (ET_a)
EF = Exposure frequency (days/yr) when a child (EF_c) or an adult (EF_a)
ED = Exposure duration (years) when a child (ED_c) or an adult (ED_a)
AT = Averaging time (days) while a child (AT_c) or an adult (AT_a)

Default values and assumptions recommended by USEPA (1989a, 1991b, 1993a) for evaluation of exposure by dermal contact with surface water are listed below. The RME intake rate for incidental water ingestion by recreational visitors of 30 mL/hour (RME) is the basis for the 10 mL/day value proposed in the Draft Water Quality Criteria Methodology Revisions (USEPA, 1998). Splashing or hand-to face contact while wading might result in only a very small amount of water in or near the mouth. For the CTE exposure scenario, the USEPA (1989a) default of 50 mL/hour for incidental ingestion during swimming is thought to be too high under this scenario. Based on this reasoning, a CTE value of 5 mL/hour (10% of the recommended default) was assumed. The exposure frequency is estimated to be 2 days per year for CTE individuals and 10 days per year for RME individuals, based on the assumption that the low intensity visitor is exposed to these media during 1 out of every 10 standard visits (4 visits per year (CTE) and 10 visits per year (RME)) and that 50% (CTE) and 100% (RME) of all visits occur at the Richardson Flats site. These exposure parameters are summarized below:

Exposure Parameters for Ingestion of Surface Water	CTE		RME	
	Child	Adult	Child	Adult
IR (mL/hour)	5	5	30	30
BW (kg)	15	70	15	70
ET (hours/day)	0.5	0.5	1.5	1.5
EF (days/year)	2	2	10	10
ED (years)	2	7	6	24
AT (non-cancer effects) (days)	2*365	7*365	6*365	24*365
AT (cancer effects) (days)	--	70*365	--	70*365

Based on these exposure parameters, the HIF values for ingestion of river water by recreational visitors are as follows:

Recreational Exposure to Surface Water	HIF (L/kg-d)	
	CTE	RME
Chronic (non-cancer)	3.6E-07	2.2E-05
Lifetime (cancer)	4.6E-08	9.6E-06

5.0 EXPOSURE ASSUMPTIONS FOR LEAD

The biokinetic slope factor approach described by Bowers et al. has been identified by EPA's Technical Workgroup for Lead as a reasonable interim methodology for assessing risks to adults from exposure to lead and for establishing risk-based concentration goals that will protect older children and adults from lead. For this reason, this method was used for estimating soil lead and tailings lead levels that could be of concern to older children and adult visitor engaging in either low-intensity or high-intensity activities at this site. When adults are exposed, the sub-population of chief concern is pregnant women and women of child-bearing age, since the blood lead level of a fetus is nearly equal to the blood lead level of the mother (Goyer 1990). Therefore, the population of concern was shifted to a slightly older (child-bearing age), female visitor.

The Bowers model predicts the blood lead level in an adult exposed to lead in a specified occupational setting by summing the "baseline" blood lead level (PbB_0) (that which would occur in the absence of any above-average site-related exposures) with the increment in blood lead that is expected as a result of increased exposure due to contact with a lead-contaminated site medium. The latter is estimated by multiplying the absorbed dose of lead from site-related exposure by a "biokinetic slope factor" (BKSF). Thus, the basic equation is:

$$PbB = PbB_0 + (PbS \cdot BKSF \cdot IR_s \cdot AF_s \cdot EF_s)/AT$$

where:

PbB = Central estimate of blood lead concentrations (ug/dL) in adults (i.e., women of child-bearing age) that have site exposures to soil lead at concentration, PbS.

PbB_0 = Typical blood lead concentration (ug/dL) in adults (i.e., women of child-bearing age) in the absence of exposures to the site that is being assessed.

BKSF = Biokinetic slope factor relating (quasi-steady state) increase in typical adult blood lead concentration to average daily lead uptake (ug/dL blood lead increase per ug/day lead uptake)

PbS = Soil lead concentration (ug/g) (appropriate average concentration for individual)

IR_s = Intake rate of soil, including both outdoor soil and indoor soil-derived dust (g/day)

AF_s = Absolute gastrointestinal absorption fraction for ingested lead in soil and lead in dust derived from soil (dimensionless). The value of AF_s is given by:

$$AF_s = AF(food) * RBA(soil)$$

EF_s = Exposure frequency for contact with assessed soils and/or dust derived in part from these soils (days of exposure during the averaging period)

AT = Averaging time; the total period during which soil contact may occur; 365 days/year for continuing long term exposures.

Once the geometric mean blood lead value is calculated, the full distribution of likely blood lead values in the population of exposed people can then be estimated by assuming the distribution is lognormal with some specified geometric standard deviation (GSD). Specifically, the 95th percentile of the predicted distribution is given by the following equation (Aitchison and Brown 1957):

$$95th = GM \cdot GSD^{1.645}$$

Input values selected for each of these parameters are summarized below:

Parameter	Low Intensity User	High Intensity User	Source
PbB_0 (ug/dL)	1.36	1.36	USEPA (2002, Table 3c) weighted average of females age 17- 45 years in the West Census Region.
PbS (ppm)	1331	1331	UCL95 Site lead concentration based on a log-normal distribution
BKSF (ug/dL per ug/day)	0.4	0.4	USEPA (1996b)
IR (g/day exposed)	0.025	0.05	Based on intake rate of 25 and 50 mg/day for low and high intensity users, respectively as discussed in Section 5. Multiplied by a factor of 1E-03 g/mg.
EF_s (days exposed at site/yr)	19.5	19.5	Based on CTE exposure assumptions for arsenic (see Section 5.1.2).
AT (days)	365	365	USEPA (1996b)
AF_o (unitless)	0.12	0.12	Based on an absorption factor for soluble lead of 0.20 (USEPA 1996b) and a relative bioavailability of 0.6
GSD	2.07	2.07	USEPA (2002, Table 3c) weighted average of females age 17- 45 years in the West Census Region.

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APPENDIX C

RBC CALCULATIONS

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RBCs were calculated for use in the COPC screening process using intake parameters for the RME exposure scenarios developed in the Exposure Assumptions document for this site (Appendix B). RBCs for sediment, surface water and soil/tailings are based on the most stringent concentration calculated for RME (high and low intensity) visitors for ingestion of each media. The RBC for air is based on inhalation of estimated airborne concentrations due to disturbance of soil/tailings. RfDs, RfCs, and slope factors used in RBC calculations are based on the Region 3 RBC Table and the online IRIS database. RBCs are based on Target Risk levels of 1E-06 for carcinogenic chemicals and a hazard quotient (HQ) of 0.1 for noncarcinogenic chemicals. Table B-1 shows all of the values used to calculate the RBC values used in the COPC selection process.

Table B-1: **RBC Calculations**

Human Intake factor
relative bioavailability
of chemical
Daily intake

Soil/Tailing

Low Intensity User

Part A: EVALUATION OF CHRONIC NONCANCER RISK

Analyte	RBC mg/kg	HIFs kg/kg-d	RBAs --	RME DIs mg/kg-d	RfD mg/kg-d	HQ --
Arsenic	71.86	5.22E-07	0.80	3.00E-05	3.00E-04	1.000E-01
Cadmium	191.63	5.22E-07	1.00	1.00E-04	1.00E-03	1.000E-01
Mercury	57.49	5.22E-07	1.00	3.00E-05	3.00E-04	1.000E-01
Silver	40379305	5.22E-07	1.00	2.11E+01	5.0E-03	1.000E-01

Part B: EVALUATION OF CANCER RISK

Analyte	RBC mg/kg	HIFs kg/kg-d	RBAs --	DIs mg/kg-d	SF	Risk --
Arsenic	3.73	2.24E-07	0.80	6.67E-07	1.50E+00	1.000E-06

High Intensity User

Part A: EVALUATION OF CHRONIC NONCANCER RISK

Analyte	RBC mg/kg	HIFs kg/kg-d	RBAs --	RME DIs mg/kg-d	RfD mg/kg-d	HQ --
Arsenic	95.81	3.91E-07	0.80	3.00E-05	3.00E-04	1.000E-01
Cadmium	255.50	3.91E-07	1.00	1.00E-04	1.00E-03	1.000E-01
Mercury	76.65	3.91E-07	1.00	3.00E-05	3.00E-04	1.000E-01
Silver	53839601	3.91E-07	1.00	2.11E+01	5.0E-03	1.000E-01

Reasonable
Maximum
Exposure

Part B: EVALUATION OF CANCER RISK

Analyte	RBC mg/kg	HIFs kg/kg-d	RBAs --	DIs mg/kg-d	SF	Risk --
Arsenic	6.21	1.34E-07	0.80	6.67E-07	1.50E+00	1.000E-06

Sediment

Part A: EVALUATION OF CHRONIC NONCANCER RISK

Analyte	RBC mg/kg	HIFs kg/kg-d	RBAs --	RME DIs mg/kg-d	RfD mg/kg-d	HQ --
Aluminum	3832463.36	2.61E-08	1.00	1.00E-01	1.00E+00	1.000E-01
Antimony	1533.00	2.61E-08	1.00	4.00E-05	4.00E-04	1.000E-01
Arsenic	1437.19	2.61E-08	0.80	3.00E-05	3.00E-04	1.000E-01
Barium	268275.00	2.61E-08	1.00	7.00E-03	7.00E-02	1.000E-01
Beryllium	7665.00	2.61E-08	1.00	2.00E-04	2.00E-03	1.000E-01
Cadmium	3832.50	2.61E-08	1.00	1.00E-04	1.00E-03	1.000E-01
Manganese	536550.00	2.61E-08	1.00	1.40E-02	1.40E-01	1.000E-01
Mercury	1149.75	2.61E-08	1.00	3.00E-05	3.00E-04	1.000E-01
Nickel	76650.00	2.61E-08	1.00	2.00E-03	2.00E-02	1.000E-01
Silver	19162.69	2.61E-08	1.00	5.00E-04	5.00E-03	1.000E-01
Thallium	306.60	2.61E-08	1.00	8.00E-06	8.00E-05	1.000E-01
Vanadium	34492.50	2.61E-08	1.00	9.00E-04	9.00E-03	1.000E-01

Part B: EVALUATION OF CANCER RISK

Analyte	RBC mg/kg	HIFs kg/kg-d	RBAs --	DIs mg/kg-d	SF	Risk --
Arsenic	74.55	1.12E-08	0.80	6.67E-07	1.50E+00	1.000E-06

WATER

Part A: EVALUATION OF CHRONIC NONCANCER RISK

Analyte	RBC mg/L	HIFs L/kg-d	RBAs --	RME DIs mg/kg-d	RfD mg/kg-d	HQ --
Aluminum	3787.878788	2.64E-05	1.00	1.00E-01	1.00E+00	1.000E-01
Ammonia	0	2.64E-05	1.00	0.00E+00		#DIV/0!
Antimony	1.515167	2.64E-05	1.00	4.00E-05	4.00E-04	1.000E-01
Arsenic	1.42	2.64E-05	0.80	3.00E-05	3.00E-04	1.000E-01
Barium	265.15	2.64E-05	1.00	7.00E-03	7.00E-02	1.000E-01
Beryllium	7.58	2.64E-05	1.00	2.00E-04	2.00E-03	1.000E-01
Boron	340.91	2.64E-05	1.00	9.00E-03	9.00E-02	1.000E-01
Cadmium	3.79	2.64E-05	1.00	1.00E-04	1.00E-03	1.000E-01

Chlorine1	378.79	2.64E-05	1.00	1.00E-02	1.00E-01	1.000E-01
Chromium VI	11.36	2.64E-05	1.00	3.00E-04	3.00E-03	1.000E-01
Cyanide	75.76	2.64E-05	1.00	2.00E-03	2.00E-02	1.000E-01
Mercury	1.14	2.64E-05	1.00	3.00E-05	3.00E-04	1.000E-01
Nickel	75.76	2.64E-05	1.00	2.00E-03	2.00E-02	1.000E-01

Part B: EVALUATION OF CANCER RISK

	RBC	**HIFs**	**RBAs**	**DIs**	**SF**	**Risk**
Analyte	**mg/L**	**L/kg-d**	**--**	**mg/kg-d**		**--**
Arsenic	0.07	1.15E-05	0.80	6.67E-07	1.50E+00	1.000E-06

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APPENDIX D

SCREENING LEVEL EVALUATION OF RELATIVE RISK FROM DERMAL CONTACT WITH SOIL

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SCREENING LEVEL EVALUATION OF RELATIVE RISK FROM DERMAL CONTACT WITH SOIL

1.0 DERMAL EXPOSURE VIA SOIL

The basic equation recommended for estimation of dermal dose from contact with soils is as follows (EPA 1989, 1992):

$$AD_{soil} = C_s \cdot SA \cdot AF \cdot ABS \cdot EF \cdot ED/(BW \cdot AT)$$

where:

C_s	=	concentration of chemical in soil (mg/kg)
SA	=	surface area in contact with soil (cm^2)
AF	=	soil adherence factor (kg/cm^2)
ABS	=	absorption fraction (unitless)

At the present time, data are very limited on the value of the ABS term, and the EPA (1992) has concluded that there are only three chemicals for which sufficient data exist to estimate credible ABS values, as shown below:

Chemical	ABS
Dioxins	0.1-3%
PCBs	0.6-6%
Cadmium	0.1-1%

It is important to realize that even these values are rather uncertain, due to a variety of differences between the exposure conditions used in laboratory studies of dermal absorption and exposure conditions that are likely to occur at Superfund sites. For example, most laboratory studies use much higher soil loadings on the skin (e.g., 5-50 mg/cm^2) than are expected to occur at sites (0.2-1 mg/cm^2). Also, most studies investigate the amount absorbed after a relatively lengthy contact period (16-96 hours), while it is expected that most people would wash off soil on the skin more promptly than this. Because of these difficulties in extrapolation from experimental measurements to "real-life" conditions, the values above are only considered approximate, and are more likely to be high than low. With respect to estimating ABS values for other chemicals (those for which there are no reliable experimental measurements), the EPA concludes that current methods are not sufficiently developed to calculate values from available data such as physical-chemical properties.

If values of ABS were available for the site COPCs, the relative magnitude of the dermal dose to the oral dose would be calculated as follows:

$$\frac{AD_d}{AD_o} = \frac{SA \cdot AF \cdot ABS \cdot EF_d}{IR \cdot AF_o \cdot EF_o}$$

where:

SA	=	surface area in contact with soil (cm^2)
AF	=	soil adherence factor (kg/cm^2)
ABS	=	absorption fraction (unitless)
IR_w	=	Ingestion rate of water (cm^3/day)
AF_o	=	Oral absorption fraction
EF_d	=	Dermal exposure frequency (days/yr)
EF_o	=	Dermal exposure frequency (days/yr)

Assuming that 10% of the body area (2,000 cm^2) is covered with soil (1 mg/cm^2 = 1E-06 kg/cm^2) for 50 days/yr, the ratio of the predicted dermal absorbed dose to the oral absorbed dose is given by:

$$\frac{AD_d}{AD_o} = 2.86 \frac{ABS}{AF_o}$$

If, by extrapolation from cadmium, the ABS is assumed to be 0.1-1% for site COPCs, then the ratio of dermal dose from soil to oral dose from soil are as follows:

Chemical	ABS (assumed)	AFo	Dose Ratio (dermal/oral)
Non-Lead COPCs	0.001-0.01	1	0.3-3%
Lead	0.001-0.01	0.1	3-28%

Because the value of ABS is not available for the site COPCs, these values should not be considered to be reliable. However, this calculation does support the conclusion that dermal absorption of metals from dermal contact with soil is likely to be relatively minor compared to the oral pathway, and omission of this pathway is not likely to lead to a substantial underestimate of exposure or risk.

4.0 REFERENCES

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APPENDIX E

ESTIMATION OF PEF VALUES

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1.0 INTRODUCTION

One pathway that humans may be exposed to contaminants in soil is by inhalation of particles of soil that become resuspended in air. When reliable site-specific measurements of contaminant levels in air due to resuspended soil particles are not available, the concentration of contaminants may be estimated as follows (USEPA 1996, 2001):

$$Cair = Csoil \cdot PEF$$

where:

Cair = Concentration of contaminant in air (mg/m^3)
Csoil = Concentration of contaminant in soil (mg/kg)
PEF = Soil to air emission factor (kg/m^3)

Note the PEF term in this equation is the inverse of the value presented in USEPA (1996, 2001), which has units of m^3/kg.

The value of PEF depends on a number of site-specific factors, as well an the nature of the force (wind, mechanical disturbance) that leads to soil particle resuspension in air. The following sections present the derivation of the PEF values used to estimate contaminant concentrations in air from the resuspension of soil attributable to wind erosion (PEF_{we}) and dirt-bike riding (PEF_{dbr}).

2.0 DERIVATION OF THE PEF FOR WIND EROSION (PEF_{we})

The basic equation used to calculate the PEF for particulates suspended in air from wind erosion is (USEPA 1996, 2001):

$$PEF_{we} = \frac{0.036 \cdot (1 - V) \cdot (U_m / U_t)^3 \cdot F(x)}{3600\ \mathrm{sec}/hr \cdot (Q / C)}$$

where:

PEF_{we} = Particulate Emission Factor for wind erosion (kg/m^3)
V = Fraction of vegetative cover (unitless)
U_m = Mean annual windspeed (m/s)
U_t = Equivalent threshold value of windspeed at 7 m (m/s)

$F(x)$	=	Function dependent on U_m/U_t derived using Cowherd et al. (1985) (unitless)
x	=	$0.886 \cdot (U_m/U_t)$
Q/C	=	Inverse of soil particle concentration in air (kg/m^3) per unit release rate (kg/m^2-sec) in the center of a square source area (g/m^2-s per kg/m^3)

The value of Q/C is given by the following (USEPA 2001):

$$Q/C_{wind} = A \cdot \exp\left[(\ln A_{source} - B)^2/C\right]$$

where:

A,B,C	=	Constants based on air dispersion modeling for specific climate zones (unitless)
A_{source}	=	Size of the site or source of contamination (acres)

The default or site-specific values and assumptions for evaluating emissions from soil due to wind erosion are summarized in Table 1. Based on these parameters, the PEF for release of soil particles into air due to wind erosion at this site is 2.92E-11 kg/m^3.

3.0 DERIVATION OF THE PEF FOR DIRT BIKE RIDING (PEF_{DBR})

The PEF value for dirt bike riding was derived according to the following general equation (USEPA 2001, Equation E-3):

$$PEF_{dbr} = \frac{J_w(dbr)}{Q/C}$$

where:

PEF_{dbr}	=	*Particulate emission factor for dirt bike riding* (kg/m^3)
$J_w(dbr)$	=	PM_{10} emission rate (g/m^2-s) due to dirt-bike riding
Q/C	=	Inverse of soil particle concentration in air (kg/m^3) per unit release rate (kg/m^2-sec) in the center of a square source area (g/m^2-s per kg/m^3)

The value of J_w is given by:

$$J_w = E10 \cdot VKT / Area$$

The value of E10 is given by (Cowherd et al. 1985)

$$E10 = 8.85 \cdot (S/10) \cdot (V/24)^{0.8} \cdot (W/7)^{0.3} \cdot (T/6)^{1.2}$$

The value of VKT is calculated as:

$$VKT = N \cdot V$$

where:

E_{10} = PM_{10} emission rate due to dirt-bike riding (kg/VKT/hr)
VKT = Vehicle kilometers traveled per hour
S = Silt content of soil (%)
V = Vehicle speed (km/hr)
W = Vehicle weight (Mg, where 1 Mg = 1,000 kg)
T = Number of tires (wheels) per vehicle
N = Number of dirt bikes riding at the same time

No adjustment was used to account for days with rain or snow (as recommended in Cowherd et al. 1985), since this form of the equation calculates emission rates during the dirt-bike riding event (rather than an annual average).

Parameters

The default values and assumptions for evaluating emissions from dirt bike riding are summarized in Table 2. Based on these parameters the PEF for release of soil particles into air due to dirt-bike riding is 9.11E-08 kg/m^3.

4.0 REFERENCES

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TABLE 1. PARAMETERS USED TO CALCULATE PEF FOR WIND EROSION

Parameter	Parameter Definition	Value	Units	Source	Notes
Q/C_{wind}	Inverse of mean concentration at center of source	--	(g/m2-s per kg/m3)	USEPA (2001)	Site-specific dispersion factor (Q/Cwind) calculated based on Appendix D (exhibit D-2) using regional climate constants and site-specific source size.
V	Fraction of vegetative cover	0.994	unitless	--	Site-specific estimate (UPCM, 2003)
Um	Mean annual windspeed	3.9	m/s	Cowherd et al. (1985)	Mean annual windspeed for Salt Lake City, Utah (Cowherd et al., 1985, Table 4-1)
Ut	Equivalent threshold value of windspeed at 7 m	11.32	m/s	USEPA (1991,1996, 2001)	Default (USEPA, 1991 and 1996), based on open terrain.
F(x)	Function dependent on Um/Ut derived using USEPA (1985, Figure 4-3 and Appendix B)	0.369	unitless	Cowherd et al. (1985)	Site-specific based on Cowherd (1985, Figure 4-3 and Appendix B), using mean annual windspeed for Salt Lake City Utah
A	Constants based on air dispersion modeling for specific climate zones	13.2559	unitless	USEPA (2001)	Zone 4, Salt Lake City, UT
B	Constants based on air dispersion modeling for specific climate zones	19.2978	unitless	USEPA (2001)	Zone 4, Salt Lake City, UT
C	Constants based on air dispersion modeling for specific climate zones	221.3379	unitless	USEPA (2001)	Zone 4, Salt Lake City, UT
A_{source}	Area extent of the site or contamination	263	acres	UPCM (2003)	Approximate size of contamination source (tailing impoundments)

TABLE 2. PARAMETERS USED TO CALCULATE PEF FOR DIRT-BIKE RIDING

Parameter	Parameter Definition	Value	Units	Source	Notes
S	Silt content of the soil (%)	15	percent (%)	Cowherd et al. (1985)	Default for rural/residential is 15%, ranging from 5-68% Cowherd et al. (1985).
V	Vehicle speed	30	km/hr	Life Systems (1993)	Assumed to be approximately 20 mph
W	Vehicle weight	0.12	Mg	Life Systems (1993)	Assumed to be 0.05 Mg (50 kg) for bike and 0.07 Mg (70 kg) for the rider
T	Number of tires (wheels) per vehicle)	2	unitless	Life Systems (1993)	Assumes 2 tires per dirt bike.
N	Number of dirt bikes	3	unitless	Life Systems (1993)	Professional judgment
A	Area over which riding occurs	8.10E+05	m^2	UPCM (2003)	Approximate area of tailing impoundments (263 acres) (UPCM, 2003).

APPENDIX F

DETAILED RISK CALCULATIONS

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Exposure Point Concentrations

Location	Medium	Chemical	Detect Frequency	Max Value	Max Hit	Min Value	GM	AM	Stdev	UCL95 Norm	UCL95 LogNorm	EPC
On-site	Sediment	Arsenic	12/12	3.1E+02	3.1E+02	1.0E+02	1.5E+02	1.6E+02	6.0E+01	1.9E+02	2.0E+02	2.0E+02
On-site	Sediment	Lead	12/12	6.5E+03	6.5E+03	1.9E+03	3.2E+03	3.5E+03	1.6E+03	4.3E+03	4.4E+03	3.5E+03
Location	Medium	Chemical	Detect Frequency	Max Value	Max Hit	Min Value	GM	AM	Stdev	UCL95 Norm	UCL95 LogNorm	EPC
On-site	Surface Water	Arsenic	99/291	7.5E-01	7.5E-01	2.5E-03	4.5E-03	8.0E-03	4.4E-02	1.2E-02	6.2E-03	1.2E-02
On-site	Surface Water	Lead	211/425	2.6E+01	2.6E+01	1.5E-03	1.0E-02	1.3E-01	1.3E+00	2.4E-01	5.3E-02	1.3E-01
Location	Medium	Chemical	Detect Frequency	Max Value	Max Hit	Min Value	GM	AM	Stdev	UCL95 Norm	UCL95 LogNorm	EPC
On-site	Soil & Tailings	Arsenic	59/64	2.4E+02	2.4E+02	2.5E+00	1.7E+01	4.1E+01	6.4E+01	5.4E+01	5.5E+01	5.5E+01
On-site	Soil & Tailings	Lead	62/62	5.9E+03	5.9E+03	1.4E+01	1.2E+02	6.6E+02	1.4E+03	9.5E+02	1.3E+03	6.6E+02

Estimated Concentrations of Arsenic in Air

LOW INTENSITY USER

Soil EPC mg/kg	PEF kg/m3	Estimated Air Conc mg/m3
5.5E+01	2.92E-11	1.62E-09

HIGH INTENSITY USER

Soil EPC mg/kg	PEF kg/m3	Estimated Air Conc mg/m3
5.5E+01	9.11E-08	5.05E-06

Intake Parameters

			Average	RME
Low Intensity Recreational User	**Soil/Tailings Ingestion**	Non-Cancer	5.4414E-08	5.21853E-07
		Cancer	6.99609E-09	2.23651E-07
	Soil/Tailings Inhalation	Non-Cancer	4.03653E-03	3.33986E-02
		Cancer	5.18982E-04	1.43137E-02
	Ingestion of Surface Water	Non-Cancer	3.5515E-07	2.23092E-05
		Cancer	4.56621E-08	9.56108E-06
High Intensity Recreational User	**Soil/Tailings Ingestion**	Non-Cancer	3.81605E-08	3.91389E-07
		Cancer	3.81605E-09	1.34191E-07
	Soil/Tailings Inhalation	Non-Cancer	2.74755E-03	2.34834E-02
		Cancer	2.74755E-04	8.05144E-03
	Dermal Contact w/ Surface Water	Non-Cancer	3.06443E-07	4.43053E-06
		Cancer	3.93999E-08	1.8988E-06
	Ingestion of Sediment	Non-Cancer	2.79046E-09	2.60926E-08
		Cancer	3.58774E-10	1.11826E-08

Toxicity Values

Soil & Tailings

Non-Cancer	***oRfD***	***Unit***	***Source***	***Effect***
Arsenic	3.0E-04	mg/kg-d	IRIS	hyperpigmentation
Cancer	**oSF**	**Unit**	**Source**	
Arsenic	1.5E+00	$(mg/kg\text{-}d)^{-1}$	IRIS	

Bioavailability factors

	Ingestion	Inhalation
Arsenic	0.80	0.80

Surface Water

Non-Cancer	**oRfD**	**Unit**	**Source**	**Effect**
Arsenic	3.0E-04	mg/kg-d	IRIS	hyperpigmentation
Cancer	**oSF**	**Unit**	**Source**	
Arsenic	1.5E+00	$(mg/kg\text{-}d)^{-1}$	IRIS	

Bioavailability factors

Arsenic	1.00

Sediment

Non-Cancer	**oRfD**	**Unit**	**Source**	**Effect**
Arsenic	3.0E-04	mg/kg-d	IRIS	hyperpigmentation
Cancer	**oSF**	**Unit**	**Source**	
Arsenic	1.5E+00	$(mg/kg\text{-}d)^{-1}$	IRIS	

Bioavailability factors

Arsenic	0.80

Air

Non-Cancer	**RfC***	**Unit**	**Source**	**Effect**
Arsenic	3.0E-04	mg/kg-d	* Oral RfD is used b/c no inhalation value available	
Cancer	**iSF**	**Unit**	**Source**	
Arsenic	1.5E+01	$(mg/kg\text{-}d)^{-1}$	IRIS	

Bioavailability factors

Arsenic	0.80

RISK CALCULATIONS FOR CHEMICALS IN SEDIMENT

INGESTION OF SEDIMENT

Part A: Noncancer Risks

		Average						RME				
	EPC	**HIF**	**RBA**	**DI**	**RfD**	**HQ**		**HIF**	**RBA**	**DI**	**RfD**	**HQ**
Analyte	**mg/kg**	**kg/kg-d**	**--**	**mg/kg-d**		**--**		**kg/kg-d**	**--**	**mg/kg-d**		**--**
Arsenic	1.98E+02	2.79E-09	0.80	4.4E-07	3.0E-04	1.5E-03		2.61E-08	0.80	4.1E-06	3.0E-04	1.4E-02
Total						1.5E-03						1.4E-02

Part B: Cancer Risks

	EPC	**HIF**	**RBA**	**DI**	**SF**	**Risk**		**HIF**	**RBA**	**DI**	**SF**	**Risk**
Analyte	**mg/kg**	**kg/kg-d**	**--**	**mg/kg-d**		**--**		**kg/kg-d**	**--**	**mg/kg-d**		**--**
Arsenic	1.98E+02	3.59E-10	0.80	5.68E-08	1.5E+00	1.1E-07		1.12E-08	0.80	1.77E-06	1.5E+00	3.3E-06
Total						*1E-07*						*3E-06*

RISK CALCULATIONS FOR CHEMICALS IN SURFACE WATER

INGESTION OF SURFACE WATER

Part A: Noncancer Risks

	Average						RME				
	EPC	HIF	RBA	DI	RfD	HQ	HIF	RBA	DI	RfD	HQ
Analyte	mg/L	L/kg-d	--	mg/kg-d		--	L/kg-d	--	mg/kg-d		--
Arsenic	1.23E-02	3.55E-07	1.00	4.4E-09	3.0E-04	1.5E-05	2.23E-05	1.00	2.7E-07	3.0E-04	9.1E-04
Total						1.5E-05					9.1E-04

Part B: Cancer Risks

	EPC	HIF	RBA	DI	SF	Risk	HIF	RBA	DI	SF	Risk
Analyte	mg/L	L/kg-d	--	mg/kg-d		--	L/kg-d	--	mg/kg-d		--
Arsenic	1.23E-02	4.57E-08	1.00	5.61E-10	1.5E+00	8.4E-10	9.56E-06	1.00	1.17E-07	1.5E+00	1.8E-07
Total						*8E-10*					*2E-07*

DERMAL CONTACT WITH SURFACE WATER

Part A: Noncancer Risks

	Average						RME				
Chemical	EPC	HIF	RBA	DI	RfD	HQ	HIF	RBA	DI	RfD	HQ
Analyte	mg/L	L/kg-d	--	mg/kg-d		--	L/kg-d	--	mg/kg-d		--
Arsenic	1.23E-02	3.06E-07	1.00	3.8E-09	3.0E-04	1.3E-05	4.4E-06	1.00	5.4E-08	3.0E-04	1.8E-04
Total						1.3E-05					1.8E-04

Part B: Cancer Risks

	EPC	HIF	RBA	DI	SF	Risk	HIF	RBA	DI	SF	Risk
Analyte	mg/L	L/kg-d	--	mg/kg-d		--	L/kg-d	--	mg/kg-d		--
Arsenic	1.23E-02	3.94E-08	1.00	4.84E-10	1.5E+00	7.3E-10	1.90E-06	1.00	2.33E-08	1.5E+00	3.5E-08
Total						*7E-10*					*3E-08*

RISK CALCULATIONS FOR CHEMICALS IN SOIL AND TAILINGS

INGESTION OF SOIL/TAIILING
Low-Intensity User

Part A: Noncancer Risks

				Average				RME			
	EPC	HIF	RBA	DI	RfD	HQ	HIF	RBA	DI	RfD	HQ
Analyte	mg/kg	kg/kg-d	--	mg/kg-d		--	kg/kg-d	--	mg/kg-d		--
Arsenic	5.54E+01	5.44E-08	0.80	2.4E-06	3.0E-04	8.0E-03	5.22E-07	0.80	2.3E-05	3.0E-04	7.7E-02
Total						*8.0E-03*					*7.7E-02*

Part B: Cancer Risks

	EPC	HIF	RBA	DI	SF	Risk	HIF	RBA	DI	SF	Risk
Analyte	mg/kg	kg/kg-d	--	mg/kg-d		--	kg/kg-d	--	mg/kg-d		--
Arsenic	5.54E+01	7.00E-09	0.80	3.10E-07	1.5E+00	5.8E-07	2.24E-07	0.80	9.92E-06	1.5E+00	1.9E-05
Total						*6E-07*					*2E-05*

INGESTION OF SOIL/TAIILING
High-Intensity User

Part A: Noncancer Risks

				Average				RME			
	EPC	HIF	RBA	DI	RfD	HQ	HIF	RBA	DI	RfD	HQ
Analyte	mg/kg	kg/kg-d	--	mg/kg-d		--	kg/kg-d	--	mg/kg-d		--
Arsenic	5.54E+01	3.82E-08	0.80	1.7E-06	3.0E-04	5.6E-03	3.91E-07	0.80	1.7E-05	3.0E-04	5.8E-02
Total						*5.6E-03*					*5.8E-02*

Part B: Cancer Risks

	EPC	HIF	RBA	DI	SF	Risk	HIF	RBA	DI	SF	Risk
Analyte	mg/kg	kg/kg-d	--	mg/kg-d		--	kg/kg-d	--	mg/kg-d		--
Arsenic	5.54E+01	3.82E-09	0.80	1.69E-07	1.5E+00	3.2E-07	1.34E-07	0.80	5.95E-06	1.5E+00	1.1E-05
Total						*3E-07*					*1E-05*

RISK CALCULATIONS FOR CHEMICALS IN AIR (FROM SOIL AND TAILINGS)

INHALATION OF AIR
Low-Intensity User

Part A: Noncancer Risks

			Average					RME			
	EPC	**HIF**	**RBA**	**DI**	**RfC**	**HQ**	**HIF**	**RBA**	**DI**	**RfC**	**HQ**
Analyte	**mg/m^3**	**m^3/kg-d**	--	**mg/kg-d**		--	**m^3/kg-d**	--	**mg/kg-d**		--
Arsenic	1.62E-09	4.04E-03	0.80	5.2E-12	3.0E-04	1.7E-08	3.34E-02	0.80	4.3E-11	3.0E-04	1.4E-07
Total						*1.7E-08*					*1.4E-07*

Part B: Cancer Risks

	EPC	**HIF**	**RBA**	**DI**	**SF**	**Risk**	**HIF**	**RBA**	**DI**	**SF**	**Risk**
Analyte	**mg/m^3**	**m^3/kg-d**	--	**mg/kg-d**		--	**m^3/kg-d**	--	**mg/kg-d**		--
Arsenic	1.62E-09	5.19E-04	0.80	6.72E-13	1.5E+01	1.3E-11	1.43E-02	0.80	1.85E-11	1.5E+01	3.5E-10
Total						*1E-11*					*3E-10*

INHALATION OF AIR
High-Intensity User

Part A: Noncancer Risks

			Average					RME			
	EPC	**HIF**	**RBA**	**DI**	**RfC**	**HQ**	**HIF**	**RBA**	**DI**	**RfC**	**HQ**
Analyte	**mg/m^3**	**m^3/kg-d**	--	**mg/kg-d**		--	**m^3/kg-d**	--	**mg/kg-d**		--
Arsenic	5.05E-06	2.75E-03	0.80	1.1E-08	3.0E-04	3.7E-05	2.35E-02	0.80	9.5E-08	3.0E-04	3.2E-04
Total						*3.7E-05*					*3.2E-04*

Part B: Cancer Risks

	EPC	**HIF**	**RBA**	**DI**	**SF**	**Risk**	**HIF**	**RBA**	**DI**	**SF**	**Risk**
Analyte	**mg/m^3**	**m^3/kg-d**	--	**mg/kg-d**		--	**m^3/kg-d**	--	**mg/kg-d**		--
Arsenic	5.05E-06	2.75E-04	0.80	1.11E-09	1.5E+01	2.1E-08	8.05E-03	0.80	3.25E-08	1.5E+01	6.1E-07
Total	1.62E-09					*2E-08*					*6E-07*

Risk Estimate Summary

Part A: Non-Cancer Risks from Arsenic

		Average	RME
Low Intensity	Sediment Ingestion	1.5E-03	1.4E-02
	Surface Water Ingestion	1.5E-05	9.1E-04
	Dermal Contact with Surface Water	1.3E-05	1.8E-04
	Low Intensity User Soil Ingestion	8.0E-03	7.7E-02
	Low Intensity User Air Inhalation	1.7E-08	1.4E-07
High Intensity	High Intensity User Soil Ingestion	5.6E-03	5.8E-02
	High Intensity User Air Inhalation	3.7E-05	3.2E-04
	Total Low Intensity User	*9.5E-03*	*9.2E-02*
	Total High Intensity User	*5.7E-03*	*5.8E-02*

Part B: Cancer Risks from Arsenic

		Average	RME
Low Intensity	✓Sediment Ingestion	1.1E-07	3.3E-06
	✓Surface Water Ingestion	8.4E-10	1.8E-07
	✓Dermal Contact with Surface Water	7.3E-10	3.5E-08
	✓Low Intensity User Soil Ingestion ✓tail	5.8E-07	1.9E-05
	Low Intensity User Air Inhalation	1.3E-11	3.5E-10
High Intensity	High Intensity User Soil Ingestion	3.2E-07	1.1E-05
	High Intensity User Air Inhalation	2.1E-08	6.1E-07
	Total Low Intensity User	*6.9E-07*	*2.2E-05*
	Total High Intensity User	*3.4E-07*	*1.2E-05*

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APPENDIX G

IEUBK MODEL

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IEUBK MODEL INPUT PARAMETERS FOR LEAD

For this site, two simulations were run using the IEUBK model. The first evaluated risks to a hypothetical nearby resident. The second simulation was used to address the risk observed when the hypothetical residential child engaged in recreational activities at the site.

Dietary Lead Intake: Values used for this site are equal to 70% of the EPA default values as follows. Rationale for the use of these values was presented in the Draft Baseline Human Health Risk Assessment for this site (EPA, 2001)

Age (years)	70% Dietary Intake (ug/day)
0-1	3.87
1-2	4.05
2-3	4.54
3-4	4.37
4-5	4.21
5-6	4.44
6-7	4.9

Geometric Standard Deviation (GSD): The GSD recommended as the default for the IEUBK model is 1.6 (USEPA 1994). However, several blood lead studies that have been performed in the Salt Lake City area have yielded GSD estimates of about 1.4 (Griffin et al., 1999b). Therefore, values of both 1.6 and 1.4 were evaluated in this assessment.

Soil Intake: Background soils were collected from areas surrounding the site. Although the samples do not represent "pristine" (not influenced by human activity) environmental levels, they are thought to be adequate to serve as a potential "off-site" residential concentration. Therefore, these background data were compiled and a value of 64 mg/kg of lead in soil, representing the log-normal UCL95 value, was utilized for residential exposure. Indoor dust concentrations were calculated using the EPA default ($C_{dust} = 0.7 * C_{yard\ soil}$). Other intake parameters for the residential scenario were kept as IEUBK model defaults.

The second scenario combined the residential parameters with those for occasional recreational visits. These visitor parameters were based on the average child who is thought to engage in recreational activities 19.5 days/year (39 days per year * 0.5 fraction contributed from site) and consume 50 mg of soil during each recreational event. Because recreational activities are not thought to occur 365 days/year, a time-weighted approach was used to derive values for input into the IEUBK model. Therefore, if the child visited a site 19.5 days/year they were exposed to their soil intake at the site on those days. For the remaining 345.5 days/year the child was assumed to be exposed at home at the concentration specified above. The concentration utilized for recreational exposure was the log-normal UCL95 of the surficial on-site soil and tailings, which was determined to be 1,331 mg/kg. The following table summarizes both intake and concentration parameters for soil/tailings. The weighted average value shows the number input into the IEUBK model for the combined residential/recreational exposure scenario.

Age		Days	Intake (mg/day)	Soil Concentration (mg/kg)
0-1	Residential	345.5	85	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**83**	**105**
1-2	Residential	345.5	135	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**130**	**90**
2-3	Residential	345.5	135	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**130**	**90**
3-4	Residential	345.5	135	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**130**	**90**
4-5	Residential	345.5	100	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**97**	**99**
5-6	Residential	345.5	90	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**88**	**103**
6-7	Residential	345.5	85	64
	Recreational	19.5	50	1331
	Weighted Average	**365**	**83**	**105**

Water Lead Concentrations and Intake Assumptions: For this analysis, lead concentrations in water and intake assumptions for each scenario were calculated according to the approach used above for soil/tailings. Residential water concentrations and intakes were set equal to the IEUBK default values. Because the intake rates (5 mL/event) and the site-specific lead concentrations (0.07 ug/L) are so low, the calculated weighted average was the same for the combined residential/recreational scenario as for the residential alone. Therefore, these values were the same in both model simulations.

Air Inhalation: Lead values for air were kept at the IEUBK default value of 0.1 ug/m^3. This is based on the observation that the maximum lead concentrations in soil/tailing (5,875 mg/kg) would result in a predicted air concentration of 0.007 ug/m^3 using a PEF of 1.16E-9 kg/m3 for low intensity activities. Because this number was lower than the default value, the default was retained in the IEUBK model.

Bioavailability: The default value of 0.60 was used for soil/tailings and sediment. This value corresponds to an absolute bioavailability of 0.30 as required for use in the IEUBK model.

Age Range: Geometric mean blood lead values were calculated for children aged 0 – 84 months.

Other Sources (Sediment Intake): Average recreational visitors are thought to be exposed to sediments approximately 2 times/year while visiting the site. During each visit, children are assumed to ingest 25 mg of sediment. Based on a log-normal 95UCL lead concentration of 4,446 mg/kg in sediments, this is expected to result in an additional 0.61 ug/day of lead on a yearly basis. Therefore, in the combined residential/recreational scenario, a value of 0.61 ug/day was added for each year of exposure. The following values were input into "other" sources in order to account for ingestion of lead in site sediments:

Age (years)	Other Intake (ug/day)
0-1	0.61
1-2	0.61
2-3	0.61
3-4	0.61
4-5	0.61
5-6	0.61
6-7	0.61

These values were obtained by multiplying the assumed intake of sediment (0.14 mg/day) by the average concentration of lead in site sediments (4,446 mg/kg) to obtain a lead intake of 0.61 ug/day.

- PUBLIC REVIEW DRAFT -

BASELINE ECOLOGICAL RISK ASSESSMENT FOR THE RICHARDSON FLAT TAILINGS SITE PARK CITY, SUMMIT COUNTY, UTAH

January 30, 2004

Prepared for the:

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Dale,
For your final review
& comment before
I send it to EPA
copy Center for
photocopying.
-Lynn

ORIGINAL

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LIST OF ACRONYMS AND ABBREVIATIONS

AQUIRE	AQUatic toxicity Information REtrieval
ATSDR	Agency for Toxic Substances and Disease Registry
COPC	Contaminant of Potential Concern
df	Dietary Fraction
E&E	Ecology & Environment, Inc.
EPC	Exposure Point Concentration
ERA	Ecological Risk Assessment
ERAGS	Ecological Risk Assessment Guidance for Superfund
HI	Hazard Index
HQ	Hazard Quotient
LC50	Lethal Concentration for 50% of the Study Organisms
LOAEL	Lowest Observed Adverse Effect Level
NOAEL	No Observed Adverse Effect Level
PEC	Probable Effects Concentration
RBA	Relative Bioavailability
RFT	Richardson Flat Tailings
RI/FS	Remedial Investigation/Feasibility Study
RMC	Resource Management Consultants
SAP	Sampling and Analysis Plan
SCM	Site Conceptual Model
SLERA	Screening Level Ecological Risk Assessment
STORET	STOrage & RETrieval
TEC	Threshold Effect Concentration
TRV	Toxicity Reference Value
UCL	Upper Confidence Limit
UPCM	United Park City Mines
USEPA	United States Environmental Protection Agency
WQM	Water Quality Monitoring

1 INTRODUCTION

1.1 Purpose

This document is a Baseline Ecological Risk Assessment (ERA) for the Richardson Flat Tailings (RFT) Site located near Park City, Utah (Figure 1-1). The purpose of the Baseline ERA is to describe the likelihood, nature, and extent of adverse effects to ecological receptors resulting from exposure to contaminants released to the environment as a result of past or present site activities. This information, along with other relevant information, is used by risk managers to decide whether remedial actions are needed to protect the environment from site-related releases. If remediation is warranted, an investigation is performed to evaluate the relative merits of a range of alternative remedial actions which might be undertaken to achieve risk management goals at the site.

1.2 Methods

This Baseline ERA was performed in accordance with current United States Environmental Protection Agency (USEPA) guidance for ecological risk assessments (USEPA 1992, 1997, 1998). The general sequence of steps used to carry out an ecological risk assessment at a Superfund site is illustrated in Figure 1-2 (USEPA 1997). It is important to realize that the eight steps shown in Figure 1-2 are not intended to represent a linear sequence of mandatory tasks. Rather, some tasks may proceed in parallel, some tasks may be performed in a phased or iterative fashion, and some tasks may be judged to be unnecessary at certain sites.

1.3 Organization

In addition to this introduction, this Baseline ERA report is organized into the following main sections.

> Section 2 - This section details the location, description, environmental setting, and history of the RFT Site.

> Section 3 - This section summarizes the data used to perform the risk assessment.

Section 4 - This section presents the ecological problem formulation, including a summary of the preliminary findings and conclusions, the site conceptual model, the presentation of assessment and measurement endpoints, and a description of the basic methods used in the assessment.

Section 5 - This section presents the ecological risk characterization for the aquatic receptors of concern, including fish and benthic macroinvertebrates.

Section 6 - This section presents the ecological risk characterization for amphibians.

Section 7 - This section presents the ecological risk characterization for wildlife receptors of concern.

Section 8 - This section provides a summary of the main uncertainties that limit confidence in the risk characterization for each of the exposure areas and classes of ecological receptors evaluated at the site.

Section 9 - This section provides citations for all data, methods, studies, and reports utilized in the Baseline ERA.

2 SITE CHARACTERIZATION

2.1 Site Location

The RFT Site is about 700 acres in size, and is located in Summit County in north-central Utah, approximately 40 miles northwest of Salt Lake City and about 1.5 miles northeast of Park City (Figure 1-1).

2.2 Site History

The site is situated in the Park City Mining District, an mineral-rich area where silver ore was mined and milled from a number of mining operations (RMC, 2001a). The site is currently owned by United Park City Mines (UPCM). UPCM is a consolidation of Silver King Coalition Mines Company and Park Utah Consolidated Mines Company, formed in 1953 (RMC, 2000a). The site was the former location of a mill that crushed ore from local silver mines, separating the silver-poor tailings from the silver-rich particles. From 1975 to 1981, tailings from the milling process were deposited via a slurry pipeline into an impoundment just east of Silver Creek. The area of the impoundment covers about 160 acres of the 700 acre property. Over the course of operations, approximately 420,000 tons of tailings were disposed of in the impoundment, resulting in a large, high-profile, cone-shaped feature. The presence of the cone-shaped feature allowed prevailing winds to cut into the tailings, and allowed the tailings to become wind-borne (RMC, 2001b). Milling operations at the site ended in 1982.

Starting in 1983, UPCM began placing soil cover on tailings outside of the impoundment. Between 1985 and 1988, UPCM also placed soil cover around the cone-shaped tailings structure inside the impoundment area at locations where it had dried out enough to support heavy equipment. The primary objective of placing the soil cover was to prevent prevailing winds from cutting into the cone-shaped tailings. By 1988, this work was completed and UPCM began a more aggressive program to cover all exposed tailings. By 1992, soil cover work was completed (RMC, 2000a). Shortly after completion, E&E (1993) conducted a soil depth survey within the impoundment and an inspection of the main embankment. For the 29 locations studied, one exhibited exposed tailings. As a result, UPCM placed additional soil in this area (RMC, 2000a). More recent soil cover surveys for the main impoundment, however, indicate that at some locations the soil cover is less than 12 inches in depth (RMC, 2001a; 2001b).

2.3 Current Site Features

The Focused Remedial Investigation/Feasibility Study (RI/FS) Workplan (RMC, 2000a) provides a detailed description of the current features of the site (see Figure 2-1). Information that is relevant to the assessment of ecological risks is summarized below.

Impoundment and Containment Dikes

The majority of the tailings at the RFT Site are contained in the impoundment basin, with a large earth embankment in place along the western edge of the Site. The "main embankment" is vegetated and is approximately 40 feet wide at the top, 800 feet long, and has a maximum height of 25 feet. A series of man-made dikes contain the tailings along the southern and eastern perimeter of the impoundment. The northern edge of the impoundment is naturally higher than the perimeter dikes.

Off-Impoundment Tailings

Additional tailings materials are present outside and to the south of the current impoundment area. During historic operations of the tailings pond, tailings accumulated in three naturally low-lying areas adjacent to the impoundment. Starting in 1983, UPCM covered these off-impoundment tailings with a low-permeability, vegetated soil cover. However, recent surveys of off-impoundment cover soils indicate that, at some locations, soil cover is thin or absent, leaving exposed surface tailings (RMC, 2001a). In addition to these off-impoundment tailings deposits, prevailing winds from the southeast carried tailings from the main impoundment and deposited them in the surrounding areas.

Diversion Ditches and Drainages

A diversion ditch system borders the north, south, and east sides of the impoundment to prevent surface water runoff from the surrounding land from entering the impoundment. Precipitation falling on the impoundment area creates a limited volume of seasonal surface water. The north diversion ditch collects snowmelt and storm water runoff from upslope, undisturbed areas north of the impoundment and carries it in an easterly direction towards origin of the south diversion ditch. An unnamed ephemeral drainage to the southeast of the impoundment also enters the south diversion ditch at this point. Additional water from spring snowmelt and storm water runoff enters the south diversion ditch from other areas lying south of the impoundment at a point near the southeast corner of the diversion ditch structure.

Site Wetlands and Pond

Water in the south diversion ditch flows from east to west and ultimately empties into Silver Creek near the north border of the Site. Before its confluence with Silver Creek, water from the south diversion ditch enters a small one acre pond (RMC, 2003). Water exiting the pond flows in a discrete channel where it mixes with flow from Silver Creek in a wetlands area below the main embankment (RMC, 2003). Near the northwestern corner of the wetlands area, Silver Creek flows into the wetland beneath the rail trail bridge. Water flow exits the wetlands area back into Silver Creek via a concrete box culvert under State Highway 248 (RMC, 2003).

Silver Creek

Silver Creek flows approximately 500 feet from the main embankment along the west edge of the Site. The headwaters of Silver Creek are comprised of three major drainages in the Upper Silver Creek Watershed; the Ontario Canyon, the Empire Canyon and Deer Valley. Flows from Ontario and Empire Canyons occur in the late spring to early summer months in response to snowmelt and rainfall, while Deer Valley flows appear to be perennial and originate from snowmelt and springs (RMC, 2000b). The major influence on water flow in Silver Creek near the RFT Site is the Pace-Homer (Dority Springs) Ditch, which derives most of its flow from groundwater (USEPA, 2001). The outflow from the Pace-Homer Ditch enters Silver Creek at several locations across the Prospector Square area. Significant riparian zones and wetlands exist near the RFT Site in areas that historically consisted of accumulated tailings piles.

2.4 Environmental Setting

The site is located in a broad valley with undeveloped rangeland. The site is about 6,570 feet above mean sea level and is characterized by a cool, dry, semi-arid climate (RMC, 2003). Meteorological stations located in Park City, Utah and Kamas, Utah estimate an annual precipitation of about 20 inches of water, an average low temperature of about 30°F, and an average high temperature of about 57°F (RMC, 2003).

In accordance with the State of Utah surface water code, the Weber River from the Stoddard diversion to its headwaters (including Silver Creek) is classified as a cold water fishery (3A) and is protected for cold water species of game fish and other cold water aquatic life, including the necessary aquatic organisms in the food chain. The RFT Site also provides possible habitat for fish, aquatic invertebrates, terrestrial plants, terrestrial invertebrates, mammals, birds, reptiles and amphibians.

2.5 Basis for Concern

Tailings released to the environment from ore milling operations generally contain metals that can, depending on the concentration and level of exposure, be toxic to ecological receptors. In accord with the eight-step process recommended by USEPA for evaluating ecological risks (see Figure 1-2), the ecological risk assessment process at this site was initiated by performing a Screening-Level Ecological Risk Assessment (SLERA) (USEPA, 2003a). The SLERA was intended to provide a preliminary evaluation of the potential for adverse effects to ecological receptors (aquatic, terrestrial, wildlife). Because a SLERA normally uses a number of simplifying assumptions and approaches and is intentionally conservative, the SLERA was not intended to support any final quantitative conclusions about the magnitude of the potential ecological risks. Based on the best data that were available at the time, the SLERA concluded that risks from site-related contaminants could not be excluded for any of these classes of ecological receptor, and identified a number of data items that would be needed to support a more detailed ecological risk evaluation.

3 DATA SUMMARY

3.1 Data Used in the SLERA

The SLERA (USEPA 2003a) provided a detailed description of the data which were used to perform the initial screening-level characterization of risks to ecological receptors at the site. In brief, data on the concentration of metals in site media (tailings, soil, surface water, groundwater, sediment) were compiled from eight sources, including: RMC (2000a), USEPA (1991), E&E (1993), USEPA (2001), RMC (2001a), RMC monthly sampling reports, UPCM monthly monitoring data, and Utah Water Quality Monitoring (WQM) data provided in STORET[1]. At the time of the SLERA (March 2002), measured tissue concentrations of metals were not available for biota (aquatic or terrestrial food items), so tissue burdens of metals were estimated using bioaccumulation models.

3.2 Data Collected Since the SLERA

Following completion of the SLERA, additional data were collected by UPCM in the site wetlands area and pond to support a more detailed and thorough evaluation of ecological impacts at the site. This included collection of additional abiotic and biotic samples and site-specific sediment toxicity testing. The field investigations were conducted in June 2003 (Phase I) and August 2003 (Phase II) (RMC, 2003). Figure 3-1 provides a summary of the locations that were sampled as part of these field investigations. Table 3-1 summarizes the number and types of data collected from each location and Appendix A provides the detailed analytical results.

In addition to the Phase I and II field investigations, UPCM continued to collect surface water monitoring data from several locations along Silver Creek, the south diversion ditch, the unnamed drainages flowing into the south diversion ditch, and ponded areas at the RFT Site (RMC 2001 - 2003). Surface water sampling stations are designated with a blue triangle in Figure 3-2.

[1] STORET = an online data STOrage and RETrieval system managed by USEPA.

Surface water samples were also routinely collected from several stations along Silver Creek as part of the Utah Water Quality Monitoring (WQM) program. Water quality monitoring data were obtained electronically from an EPA STORET download query (Modernized Version) performed November 13, 2003. The Silver Creek WQM stations selected for use in the Baseline ERA are described in the table below.

Station	Location Description	Latitude	Longitude	Sampling Dates
492674	Silver Creek at Farm Crossing in Atkinson	40.742167	-111.474167	12-Jan-68 to 2-Oct-03
492675	Silver Creek at Wanship above confluence with Weber River	40.813000	-111.401667	20-Dec-79 to 1-Oct-03
492676	Silver Creek 2 miles north of Atkinson	40.768500	-111.467667	21-Aug-81 to 11-May-89
492677	Silver Creek at I-80 Crossing at Atkinson east of Silver Creek Junction	40.743833	-111.473000	20-Dec-79 to 22-Jan-92
492679	Silver Creek at Waste Water Treatment Plant	40.735167	-111.474667	4-Jun-87 to 2-Oct-03
492680	Silver Creek above Atkinson	40.735167	-111.475167	17-Sep-81 to 2-Oct-03
492685	Silver Creek at US40 Crossing east of Park City	40.683000	-111.456000	2-May-75 to 2-Oct-03
492694	Silver Creek at Railroad Crossing below Park City above Landfill	40.658000	-111.501833	20-Dec-79 to 28-Nov-83
492695	Silver Creek at City Park above Prospector Square	40.654333	-111.501667	6-Aug-97 to 2-Oct-03

3.3 Exposure Areas

For the purposes of this assessment, the site was dived into a number of areas of potential ecological concern. In addition, several locations that are not believed to be impacted by site-related releases were identified to serve as reference areas for the site. These exposure areas are described in Table 3-2. Figure 2-1 provides a map of the site exposure areas and Figures 3-3 and 3-4 show the reference (background) areas sampled in aquatic and terrestrial habitats, respectively.

3.4 Summary Statistics for Environmental Media

All relevant and reliable data for the site were assembled into an electronic database (Microsoft Access®). This database is available upon request from USEPA Region 8.

Appendix B provides summary statistics (detection frequency, average, standard deviation, minimum, maximum) for each analyte in each medium for each exposure area.

4 PROBLEM FORMULATION

Problem formulation is a systematic planning step that identifies the major concerns and issues to be considered in the Baseline ERA, and a description of the basic approach that will be used to characterize the potential risks that may exist (USEPA 1997). As discussed in USEPA guidance (USEPA 1997), problem formulation is an iterative process, undergoing refinement as new information and findings become available. In accordance with this guidance, problem formulation for this ecological risk assessment began with the SLERA that was completed for the site in March 2002 (USEPA, 2003a). The following section summarizes the main findings of the SLERA, which in turn helped refine the problem formulation for the Baseline ERA.

4.1 Summary of the Screening-Level Ecological Risk Assessment (SLERA)

<u>Ecological Receptors of Potential Concern</u>

Ecological receptors evaluated in the SLERA included aquatic/semi-aquatic receptors (fish, benthic invertebrates, amphibians) in the site diversion ditches and Silver Creek, terrestrial receptors (plants, soil invertebrates) in contact with surface soils on and off the main impoundment, and wildlife receptors that reside at the site or along Silver Creek.

<u>Exposure Pathways Evaluated</u>

Exposure pathways that were quantitatively evaluated in the SLERA included:

- Direct contact of aquatic receptors (fish, benthic invertebrates, amphibians) with surface water and seep water
- Direct contact of benthic invertebrates with sediment
- Direct contact of terrestrial plants and soil invertebrates with soil and tailings
- Direct contact of terrestrial plants with seep water
- Ingestion of surface water, seep water, sediment, and soil by birds and mammals
- Ingestion of food items (fish, benthic invertebrates, terrestrial plants, soil invertebrates, small mammals) by birds and mammals

Summary of Screening-Level Risk Findings

Table 4-1 provides a summary of the screening level risk findings presented in the SLERA. Based on the preliminary risk characterization in the SLERA, further evaluation was recommended for almost all quantitative exposure pathways. No further evaluation of wildlife exposures from ingestion of surface water and seep water was recommended because predicted risks were below a level of concern.

Summary of Data Gaps

The SLERA identified a number of data areas where additional information was needed to help improve the reliability and accuracy of the risk assessment. Table 4-2 provides a summary of these data gaps and recommendations for data collection activities. These data gaps were considered in the development of a field investigation Sampling and Analysis Plan (SAP) (RMC, 2003).

4.2 Site Conceptual Model for the Baseline ERA

Figure 4-1 presents the site conceptual model (SCM) for the Baseline ERA. Because few pathways could be excluded as a result of the SLERA, this site model is very similar to the site model that was developed for the SLERA.

As indicated in the SCM, ecological receptors that may be exposed at the site include aquatic receptors (fish and benthic macroinvertebrates), amphibians and reptiles, terrestrial receptors (plants and soil invertebrates), and wildlife receptors (birds and mammals). Each receptor class may be exposed to chemical contamination via contact with one or more environmental media, including surface water, sediment, seeps, aquatic food items, soil/tailings, and terrestrial food items. However, not all of these exposure pathways are likely to be of equal concern. For the purposes of this risk assessment, each complete exposure pathway was classified as follows:

- The pathway is considered to be of potential concern, and sufficient data exist to support a quantitative risk evaluation. These cases are indicated by boxes containing a solid circle (●). These pathways are the primary focus of this risk assessment.

- The pathway is considered to be of potential concern, but available data are too limited to support a reliable quantitative risk evaluation. These cases are shown by boxes with an open circle (O).

- The risk posed by the pathway is likely to be minor, either on an absolute basis and/or in comparison to other exposure pathways that affect the same receptor. These cases are indicated by boxes with an "X". Because these pathways are judged to be of minor concern, they are not evaluated quantitatively.

- The pathway is considered to be incomplete (i.e., not thought to occur). These cases are shown as open boxes.

The following section provides a more detailed discussion of the exposure pathways selected for quantitative evaluation in the Baseline ERA.

Aquatic Receptors

- The main pathways of exposure for fish and benthic invertebrates are direct contact with surface water and sediment. Each of these pathways were evaluated quantitatively.

- Most fish have relatively low direct contact with sediment, and concern over this pathway is generally minor. Therefore, direct contact with sediment was not evaluated quantitatively for fish. This pathway was evaluated quantitatively for benthic invertebrates.

- Ingestion of aquatic food web items is a pathway of potential concern for fish and benthic invertebrates. Likewise, incidental ingestion of sediment by these receptors might occur in some cases. Although limited data are available to estimate oral TRVs for fish, exposure of fish by the oral pathway is usually thought to be of lesser importance than direct contact with surface water. Therefore, ingestion exposures were not evaluated quantitatively for fish. For benthic invertebrates, sediment based TRVs and sediment toxicity studies are likely to capture exposure both by contact with the sediment and ingestion of detritus and sediment particles, so oral exposure of invertebrates was not considered separately.

Amphibians and Reptiles

- Amphibians and reptiles may be exposed to site-related contaminants in surface water, sediment, soil, and the diet. Although these exposure pathways may be significant, data on exposure and toxicity needed to perform a quantitative evaluation for each potential exposure pathway are very limited. Of the required data, only screening-level toxicity values for direct contact exposure of amphibians to surface water could be located. Therefore, this pathway was evaluated quantitatively, while other pathways were evaluated qualitatively.

Terrestrial Receptors (Plants and Invertebrates)

- The primary exposure pathway for both terrestrial plants and soil invertebrates is direct contact with contaminated soils. For terrestrial plants, exposure may also occur due to deposition of dust on foliar (leaf) surfaces, but this pathway is believed to be small compared to root exposures. Although direct contact exposures are complete and of potential concern (USEPA, 2003a), no new data are available for contaminant concentrations in soil, and it is expected that remedial actions planned for the site will largely address potential risks to plants and soil invertebrates from soils on the main impoundment and in off-impoundment areas (RMC, 2003). Therefore, risks to terrestrial receptors were not re-evaluated in the Baseline ERA.

Wildlife Receptors (Birds and Mammals)

- Birds and mammals may be exposed by ingestion of surface water, and this pathway was evaluated quantitatively. Although the results of the SLERA indicated risks from ingestion of surface water are likely to be below a level of concern, this conclusion was based on limited data from the wetlands area. Because new data are now available, this pathway was selected for re-evaluation in the Baseline ERA to confirm that predicted risks are low.

- Inhalation exposure to airborne dusts is possible for all birds and mammals. However, this pathway is generally considered to be minor compared to ingestion pathways (USEPA, 2003b), and was not evaluated quantitatively.

- Direct contact (i.e., dermal exposure) of birds and mammals to soils, sediments, and surface water may occur in some cases, but these exposures are judged to be minor in comparison to exposures from ingestion (USEPA, 2003b), and were not evaluated quantitatively.

- Birds and mammals may be exposed by ingestion of food web items (either from the terrestrial environment and/or from the aquatic environment). Wildlife receptors may also ingest soil or sediment during feeding, especially for soil- or sediment-dwelling prey items. Although these exposure pathways are complete and of potential concern (USEPA, 2003a), no new data are available for contaminant concentrations in soil or in terrestrial food items, and it is expected that remedial actions planned for the site will largely address potential risks to terrestrial (upland) wildlife receptors from exposures to contaminants on the main impoundment and in off-impoundment areas (RMC, 2003). Therefore, quantitative risk characterization for the Baseline ERA focused on exposures of aquatic/semi-aquatic wildlife receptors in the wetlands area, and risks to upland terrestrial wildlife receptors were not re-evaluated in the Baseline ERA.

4.3 Ecological Management Goals

The overall management goal for ecological health at the RFT Site is (USEPA, 2003a):

> *Ensure adequate protection of ecological systems within the impacted areas of the Richardson Flat Tailings Site by protecting them from the deleterious effects of acute and chronic exposures to site-related contaminants of concern.*

In order to provide specificity regarding this general goal and identify specific measurable ecological values to be protected, the following list of sub-goals was derived (USEPA, 2003a):

- Ensure adequate protection of terrestrial soil fauna and plant communities, including native plant communities, by protecting them from the deleterious effects of acute and chronic exposures to site-related contaminants of concern.

- Ensure adequate protection of aquatic and amphibian life in Silver Creek, the site diversion ditches and wetlands areas from the deleterious effects of acute and chronic exposures to site-related contaminants of concern.

- Ensure adequate protection of terrestrial mammal and bird populations by protecting them from the deleterious effects of acute and chronic exposures to site-related contaminants of concern.

- Ensure adequate protection of threatened and endangered species (including candidate species) and species of special concern and their habitat by protecting them from the deleterious effects of acute and chronic exposures to site-related contaminants of concern.

"Adequate" protection is defined as protective of growth, reproduction, and survival of local populations.

4.4 Assessment and Measurement Endpoints

Assessment endpoints are explicit statements of the characteristics of the ecological system that are to be protected. Assessment endpoints are either measured directly or are evaluated through indirect measures. Measurement endpoints represent quantifiable ecological characteristics that can be measured, interpreted, and related to the valued ecological components chosen as the assessment endpoints (USEPA 1992, 1997).

Table 4-3 presents the assessment and measurement endpoints used to interpret potential ecological risks for the RFT Site that were evaluated in the Baseline ERA. These measurement endpoints can be divided into three basic categories of approach, as follows:

- Hazard Quotients (HQs)
- Site-specific toxicity tests
- Observations of population and community demographics

These three basic approaches are described in more detail below.

4.4.1 Hazard Quotients

A Hazard Quotient (HQ) is the ratio of the estimated exposure of a receptor at the site to a "benchmark" exposure that is believed to be without significant risk of unacceptable adverse effect:

HQ = Exposure / Benchmark

Exposure may be expressed in a variety of ways, including:

- Concentration in an environmental medium (water, sediment, soil, diet)
- Concentration in the tissues of an exposed receptor
- Amount of chemical ingested by a receptor

In all cases, the benchmark toxicity value must be of the same type as the exposure estimate.

If the value of an HQ is less than or equal to 1, risk of unacceptable adverse effects in the exposed individual is judged to be acceptable. If the HQ exceeds 1, the risk of adverse effect in the exposed individual is of potential concern.

When interpreting HQ results for ecological receptors, it is important to remember that the assessment endpoint is usually based on the sustainability of exposed populations, and risks to some individuals in a population may be acceptable if the population is expected to remain healthy and stable. In these cases, population risk is best characterized by quantifying the fraction of all individuals that have HQ values greater than 1, and by the magnitude of the exceedences.

The fraction of the population that must have HQ values below a value of 1 in order for the population to remain stable depends on the species being evaluated and on the toxicological endpoint underlying the toxicity benchmark, and reliable characterization of the impact of a chemical stressor on an exposed population risks requires knowledge of population size, birth rates, and death rates, as well as immigration and emigration rates. Because this type of detailed knowledge of population dynamics is generally not available on a site-specific basis, extrapolation from a distribution of individual risks to a characterization of population-level risks is generally uncertain. However, if all or nearly all of the HQs for individuals in a population of receptors are below 1, it is very unlikely that unacceptable population-level effects will occur in the exposed population. Conversely, if many or all of the individual receptors have HQs that are above 1, then unacceptable effects on the exposed population are likely, especially if the HQ values are large. If only a small portion of the exposed population has HQ values that exceed 1, some individuals may be impacted, but population-level effects are not likely to occur. As the fraction of the population with HQ values above 1 increases, and as the magnitude of the exceedences increases, risk that a population-level effect will occur also increases. This concept is illustrated schematically in Figure 4-2.

In practice, estimating the distribution of HQ values in different individuals in a population is not always easy. Variability in the HQ for different members of a population can arise from one or both of two sources, depending on the size of the exposure area being assessed and the size of the home range of the receptor of concern. In cases where the home range is as large as the exposure area, and assuming the receptors tend to be exposed at random across the exposure area, exposure is related to the mean concentration across the exposure area (this is a constant, not a variable), and variation in exposure is related mainly to differences in the intake rates (dietary fractions) of different environmental media. For receptors that have a small home range compared to the size of the exposure area, the population consists of individuals residing at a number of different home ranges within the exposure area, and variability in the mean concentration of contaminant across different home ranges is usually the primary reason for between-individual variation in exposure.

Based on this, variability in exposure among individuals with small home ranges (this includes plants, soil invertebrates, many small mammals and birds, benthic macroinvertebrates, and many fish) can be approximated by the variability in concentration values at different locations in the exposure area. It is important to note that this is only an approximation, since population density is often not uniform across an exposure area, depending on a number of key habitat variables. Thus, if 20% of all sampling locations in an exposure area yielded an HQ above 1, it is reasonable to estimate that about 20% of the population of small home range receptors could be at risk, but the actual fraction could be either lower or higher, depending on variability in habitat suitability.

Several other factors can also make calculation or interpretation of HQ distributions difficult. First, if the number of data points collected from an exposure area is small and if the data points tend to lie near a value of 1, it is difficult to estimate the fraction of HQs above 1 with certainty. For example, an exceedence frequency of 33% based on 10 exceedences out of 30 samples is much more reliable than 1 exceedence out of 3 samples. Second, when a substantial fraction of the available concentration data are below the limit of detection, and the limit of detection is above the level corresponding to an HQ of 1, it is usually not possible to draw a firm conclusion, since HQ values for non-detects might be either above or below a level of 1. Third, in cases where the HQ values for a reference area are above 1, it is difficult to interpret the predicted risk of adverse effects in site areas, since elevated HQ values are not expected for reference areas, suggesting that the benchmarks used to calculate HQs may be somewhat over-protective. However, the degree to which the benchmark is over-protective is unknown, and hence the true risks to receptors in site exposure areas are also uncertain.

It interpreting HQ values and distributions of HQ values, it is always important to bear in mind that the values are predictions, and are subject to the uncertainties that are inherent in both the estimates of exposure and the estimates of toxicity benchmarks. Therefore, HQ values should be interpreted as estimates rather than highly precise values, and should be viewed as part of the weight-of-evidence along with the results of site-specific toxicity testing and direct observations on the structure and function of the aquatic community (see below).

4.4.2 Site-Specific Toxicity Tests

Site-specific toxicity tests measure the response of receptors that are exposed to site media. This may be done either in the field or in the laboratory using media collected on the site. The chief advantage of this approach is that site-specific conditions which can influence toxicity are usually accounted for. A potential disadvantage is that, if toxic effects are observed to occur when test organisms are exposed to a site medium, it is usually not possible to specify which chemical or combination of chemicals is responsible for the effect. Rather, the results of the toxicity testing reflect the combined effect of the mixture of chemicals present in the site medium. In addition, it is often difficult to test the full range of environmental conditions which may occur at the site across time and space, either in the field or in the laboratory, so these studies are not always adequate to identify the boundary between exposures that are acceptable and those that are not.

4.4.3 Population and Community Demographic Observations

A third approach for evaluating impacts of environmental contamination on ecological receptors is to make direct observations on the receptors in the field, seeking to determine whether any receptor population has unusual numbers of individuals (either lower or higher than expected), or whether the diversity (number of different species) of a particular category of receptors (e.g., plants, benthic organisms, small mammals, birds) is different than expected. The chief advantage of this approach is that direct observation of community status does not require making the numerous assumptions and estimates needed in the HQ approach. However, there are also a number of important limitations to this approach. The most important of these is that both the abundance and diversity of an ecological population depend on many site-specific factors (habitat suitability, availability of food, predator pressure, natural population cycles, meteorological conditions, etc.), and it is often difficult to know what the expected (non-impacted) abundance and diversity of an ecological population should be in a particular area. This problem is generally approached by seeking an appropriate "reference area" (either the site itself before the impact occurred, or some similar site that has not been impacted), and comparing the observed abundance and diversity in the reference area to that for the site. However, it is sometimes quite difficult to locate reference areas that are truly a good match for all of the important habitat variables at the site, so comparisons based on this approach do not always establish firm cause-and-effect conclusions regarding the impact of environmental contamination on a receptor population.

4.4.4 Weight of Evidence Evaluation

As noted above, each of the measurement endpoints has advantages but also has limitations. For this reason, conclusions based on only one method of evaluation may be misleading. Therefore, the best approach for deriving reliable conclusions is to combine the findings across all of the methods for which data are available, taking the relative strengths and weaknesses of each method into account. If the methods all yield similar conclusions, confidence in the conclusion is greatly increased. If different methods yield different conclusions, then a careful review must be performed to identify the basis of the discrepancy, and to decide which approach provides the most reliable information.

5 EVALUATION OF RISKS TO AQUATIC RECEPTORS

As discussed above, aquatic receptors (fish, benthic invertebrates) may be exposed to site contaminants in surface water and sediment at a number of exposure areas including Silver Creek, the south diversion ditch, the wetlands area, the site pond, and an unnamed drainage which flows into the south diversion ditch (Figure 2-1). Evaluation of potential risks by the Hazard Quotient (HQ) approach, site-specific toxicity testing, and population surveys are summarized below.

5.1 HQ Approach: Direct Contact with Surface Water

For surface water exposures, the HQ is given by the ratio of the measured surface water concentration divided by an appropriate surface water toxicity benchmark:

$$HQ = \text{Exposure Conc}_{sw} / \text{Benchmark}_{sw}$$

where:

Conc_{sw} = chemical concentration in surface water (ug/L)
Benchmark_{sw} = chemical toxicity benchmark for surface water (ug/L)

The following sections describe the inputs used to calculate HQs and summarize the predicted risks to aquatic receptors from surface water.

5.1.1 Exposure Assessment

Because concentrations of chemicals in surface water can vary significantly over time and location, exposure of aquatic receptors is best characterized as a distribution of individual values at each sampling location, rather than as an average of values over time and/or over location. Therefore, HQs were calculated for each sample for each chemical. In accord with USEPA guidance, non-detects were evaluated at one-half the detection limit.

For inorganics, concentration values in surface water may be expressed either as total recoverable or as "dissolved" (that which passes through a fine-pore filter). There is general consensus that toxicity to aquatic receptors is dominated by the level of dissolved chemicals

(Prothro, 1993), since chemicals that are adsorbed onto particulate matter may be less toxic than the dissolved forms. Therefore, aquatic receptor exposures to inorganics in surface water were evaluated using dissolved concentrations.

Appendix B provides summary statistics (detection frequency, average, standard deviation, minimum, maximum) for each analyte in surface water for each exposure area. Raw surface water data are provided electronically in the site database.

5.1.2 Toxicity Assessment

Toxicity benchmark values for the protection of aquatic life from direct contact with chemicals in surface water are available from several sources. These toxicity values are designed to be protective of fish, benthic invertebrates, and some aquatic plants. Each of the sources evaluated in deriving surface water toxicity benchmarks is described briefly in Appendix C. This appendix also describes the hierarchy used to identify the most relevant and reliable toxicity benchmark value when more than one value was available.

Two different types of toxicity benchmark were selected. The acute toxicity benchmark is intended to protect against short-term (48-96 hour) lethality, while the chronic toxicity benchmark is intended to protect against long-term effects on growth, reproduction, and survival. Table 5-1 presents the acute and chronic toxicity benchmark values for all chemicals detected in surface water at the site. In cases where the toxicity values are dependent on the hardness of the water, benchmark values were calculated for each water sample using the measured hardness in the sample and the hardness-dependant equations provided in USEPA (2002a). If the hardness value for a sample was not reported, a value equal to the average hardness of other samples from the same location was assumed. In accord with guidance, if the observed hardness exceeded 400 mg/L (e.g., upstream Silver Creek, site diversion ditch), the hardness value was assumed to be 400 mg/L (USEPA 2002a). Hardness-dependant values shown in Table 5-1 are based on a hardness of 85 mg/L (the lowest hardness observed at the site).

5.1.3 Selection of COPCs

Although Chemicals of Potential Concern (COPCs) were selected for this exposure pathway previously in the SLERA, additional surface water data have been collected since the SLERA. Therefore, surface water COPCs were re-evaluated in this assessment incorporating the new surface water results.

The general procedure used to identify COPCs is provided in Figure 5-1. This procedure is based on conservative estimates of exposure and toxicity to ensure that all chemicals that may be of potential concern will be carried forward for further quantitative evaluation.

For the purposes of selecting COPCs for surface water, the maximum detected dissolved concentration of each chemical was compared to the chronic toxicity benchmark for that chemical, assuming a hardness equal to the lowest hardness measured at the site (85 mg/L). If the maximum detected concentration was greater than the most conservative toxicity benchmark, the chemical was retained for further quantitative evaluation. Table 5-2 provides the results of the COPC selection for aquatic receptors from direct contact with surface water.

5.1.4 Risk Characterization

Appendix D presents the detailed calculations of HQ values for each quantitative COPC in each surface water sample, along with graphs which summarize the distributions of HQ values for samples collected at each exposure area. The HQ distribution graphs presented in Appendix D were evaluated using the semi-quantitative approach for interpretation of HQs described in Section 4.4.1 and illustrated graphically in Figure 4-2. Figure 5-2 and 5-3 provide examples of calculated surface water HQs for cadmium and zinc, respectively. In each figure, the upper panel shows the distribution of HQ values for acute toxicity, while the lower panel reflects the distribution of risks of chronic effects on growth or reproduction. HQs based on non-detects are shown as open-circles and HQs based on detects are shown as closed circles. Note that the results in these figures are plotted on a log-scale, so large differences between HQ values are somewhat compressed.

Table 5-3 summarizes the conclusions drawn from the graphs, based on a consideration of the number of exceedences (HQ > 1), the magnitude of the exceedences, the number of data points, and a comparison of site values to reference areas. Inspection of this table suggests the following main findings:

- Data are generally not sufficient to draw a firm conclusion regarding risks from boron, calcium, manganese and magnesium. However, it is not considered likely that any of these chemicals are major risk drivers at the site.

- Most other chemicals in surface water do not have any HQ values greater than 1, or else have a relatively low frequency and magnitude of exceedences, suggesting that risk of population-level effects to aquatic receptors is low. However, cadmium and zinc do have HQ distributions that suggest risks may be of concern at several locations, as discussed below.

- In Silver Creek, zinc and cadmium appear to pose moderate to high risk both upstream and downstream of the RFT site. Because the risks are high in water upstream of the site, it is evident that much of the risk is attributable to chemicals derived from upstream sources, and the contribution from the site is difficult to observe.

- Zinc appears to pose a moderate risk to aquatic receptors in the site diversion ditch and the unnamed drainages. It is likely that most of the zinc in the ditch and the drainages is site-related.

- Risks do not appear to be of substantial concern from contaminants in surface water in the wetlands area and the site pond.

5.2 HQ Approach: Direct Contact with Sediment

Risks to benthic invertebrates from direct contact with sediment may be evaluated either using measured concentrations in bulk sediment, or using measured concentrations in sediment porewater. Each of these methods are described below.

5.2.1 HQ Values Based on Bulk Sediment Measurements

In this evaluation, the HQ is the ratio of the measured bulk sediment concentrations to an appropriate bulk sediment toxicity benchmark, as follows:

$$HQ = Conc_{sed} / Benchmark_{sed}$$

where:

$Conc_{sed}$ = chemical concentration in bulk sediment (mg/kg)
$Benchmark_{sed}$ = chemical toxicity benchmark for bulk sediment (mg/kg)

The following sections describe the inputs used to calculate HQs and summarize the predicted risks to benthic invertebrates from bulk sediment.

5.2.1.1 Exposure Assessment

Benthic macroinvertebrates that spend some or most of their life cycle within the sediment substrate are exposed to chemicals through direct contact with sediment. Although concentrations of chemicals in sediment are usually not as time-variable as concentrations in surface water, concentrations do fluctuate as contaminated material is added or removed by surface water flow. In addition, there may be significant small scale variability in sediment concentrations at any specific sampling station. Therefore, exposure to sediments is usually best characterized as a distribution of individual values at a specific exposure area.

Appendix B provides summary statistics (detection frequency, average, standard deviation, minimum, maximum) for each analyte in bulk sediment for each exposure area. Raw bulk sediment data are provided electronically in the site database.

5.2.1.2 Toxicity Assessment

Toxicity benchmark values for the protection of benthic invertebrates from direct contact with chemicals in bulk sediment are available from several sources. Each of the sources evaluated in deriving sediment toxicity benchmarks is described briefly in Appendix C. This appendix also describes the hierarchy used to identify the most relevant and reliable toxicity benchmark value when more than one value was available. The selected sediment toxicity benchmark is the concentration below which toxicity is expected to occur only rarely. This level is referred to as the threshold effect concentration (TEC)[2]. Table 5-4 presents the toxicity benchmark values for benthic invertebrates from direct contact with bulk sediment.

[2] The TEC encompasses several types of sediment quality guidelines including the Lowest Effect Level (LEL), the Threshold Effect Level (TEL), the Effect Range Low (ERL), and the Minimum Effect Threshold (MET). See Appendix C for a detailed description of each of these terms.

5.2.1.3 Selection of COPCs

Although COPCs were previously selected for this exposure pathway in the SLERA, additional sediment data have been collected since the SLERA. Therefore, sediment COPCs were re-evaluated in this assessment incorporating the new sediment results.

The general procedure shown in Figure 5-1 was used to identify the COPCs for bulk sediment. Maximum detected concentrations in sediment were compared to their respective TEC benchmark. If the maximum detected concentration was greater than the toxicity benchmark, the chemical was retained for further quantitative evaluation.

Table 5-5 provides the results of the COPC selection for aquatic receptors from direct contact with bulk sediment.

5.2.1.4 Risk Characterization

Appendix E presents the detailed calculations for each quantitative COPC in each bulk sediment sample, along with graphs which summarize the distributions of HQ values for samples collected at each exposure area. The HQ distribution graphs presented in Appendix E were evaluated using the semi-quantitative approach for interpretation of HQs described in Section 4.4.1 and illustrated graphically in Figure 4-2. Figure 5-4 provides the HQ distribution graphs for cadmium and copper. HQs based on non-detects are shown as open-circles and HQs based on detects are shown as closed circles. Note that the results in these figures are plotted on a log-scale, so large differences between HQ values are somewhat compressed.

Table 5-6 summarizes the conclusions drawn from the graphs, based on a consideration of the number of exceedences (HQ > 1), the magnitude of the exceedences, the number of data points, and a comparison of site values to reference areas. Inspection of this table suggests the following main findings:

- Cadmium, copper and mercury appear to be of concern in all on-site locations. Nickle may also be of concern in the wetlands area, and silver and zinc may be of concern in the site pond.

- In Silver Creek, risks from sediment appear to be similar in both upstream and downstream locations, indicating that most of the estimated risk is attributable to chemicals derived from upstream sources. Contaminants in site diversion ditch, wetlands are site pond are more likely to be site-related.

- Risks from antimony, arsenic, lead, manganese, silver, and zinc were difficult to interpret because HQs in the reference areas were above a level of concern which suggests that the toxicity benchmarks for these chemicals are over-predicting risks from metals in bulk sediment. The degree to which the benchmark is over-protective is unknown, therefore, the true risks to receptors in site exposure areas are also uncertain.

5.2.2 HQ Approach Based on Sediment Porewater Measurements

Adverse effects from direct contact with sediment are likely to be mediated primarily by chemicals that have dissolved into sediment porewater from the bulk sediment. Thus, the most direct approach for evaluating toxicity from chemicals in sediment is to measure the concentrations in the sediment porewater and compare those concentrations to water-based toxicity values. For porewater, the HQ is the ratio of the measured porewater concentration to an appropriate water toxicity benchmark, as follows:

$$HQ = \text{Exposure Conc}_{pw} / \text{Benchmark}_{pw}$$

where:

Conc_{pw} = chemical concentration in sediment porewater (ug/L)
Benchmark_{pw} = chemical toxicity benchmark for water (ug/L)

The following sections describe the inputs used to calculate HQs and summarize the predicted risks to benthic invertebrates from sediment porewater.

5.2.2.1 Exposure Assessment

Since there may be both spatial and temporal variability in sediment porewater concentrations at any specific sampling station, exposure to benthic macroinvertebrates is usually best characterized as a distribution of concentration values at a specific location. However, at this site, there is only one measurement of porewater available per sampling location, so exposure was based on that single concentration value. As noted above, because toxicity to aquatic receptors from water exposure is dominated by the level of dissolved chemicals, exposures to inorganics in sediment porewater were evaluated using dissolved concentrations.

Appendix B provides summary statistics (detection frequency, average, standard deviation, minimum, maximum) for each analyte in sediment porewater for each exposure area. Raw sediment porewater data are provided electronically in the site database.

5.2.2.2 Toxicity Assessment

The acute and chronic toxicity benchmark values for surface water described in Section 5.1.2 (above) were used to evaluate potential risks to benthic invertebrates from direct contact with sediment porewater. Although these toxicity values are designed to be protective of fish, benthic invertebrates, and some aquatic plants, they were used without adjustment for the purposes of screening risks to benthic organisms from sediment porewater.

5.2.2.3 Selection of COPCs

The general procedure shown in Figure 5-1 was used to identify the COPCs for sediment porewater. Maximum detected dissolved concentrations in sediment porewater were compared to their respective long-term chronic toxicity benchmark based on the lowest hardness measured in sediment porewater samples (351 mg/L). If the maximum detected concentration was greater than the most conservative toxicity benchmark, the chemical was retained for further quantitative evaluation. Table 5-7 provides the results of the COPC selection for benthic invertebrates from direct contact with sediment porewater.

5.2.2.4 Risk Characterization

Appendix F presents the detailed calculations of HQ values for each quantitative COPC in each sediment porewater sample, along with graphs which summarize the distributions of HQ values for samples collected at each exposure reach. Two examples are presented in Figure 5-5 (arsenic) and Figure 5-6 (zinc). In each figure, the upper panel shows the distribution of HQ values for based on the acute toxicity benchmark, while the lower panel reflects the distribution of risks of based on the chronic toxicity benchmark. HQs based on non-detects are shown as open-circles and HQs based on detects are shown as closed circles. Note that the results in these figures are plotted on a log-scale, so large differences between HQ values are somewhat compressed.

The HQ distribution graphs presented in Appendix F were evaluated using the semi-quantitative approach for interpretation of HQs described in Section 4.4.1 and illustrated graphically in Figure 4-2. Table 5-8 summarizes the conclusions drawn from the graphs, based on a consideration of the number of exceedences (HQ > 1), the magnitude of the exceedences, the number of data points, and a comparison of site values to reference areas. Inspection of this table suggests the following main findings:

- In the wetlands area, arsenic and zinc appear to be of potential concern for both acute and chronic toxicity, while antimony, cadmium, and lead appear to be of chronic (but not acute) concern. The locations that yield the highest risk estimates are generally from the northern portion of the wetlands (SD-2, SD-4, SD-6; see Figure 3-1).

- Risks do not appear to be of substantial concern from contaminants in sediment porewater in the site pond.

5.3 Site-Specific Toxicity Tests

One way to help reduce the uncertainty associated with risk predictions based on the HQ approach is to perform direct toxicity testing using site-specific media. Tests of this type have been performed to investigate the toxicity of site sediments on benthic organisms, using sediment samples collected from the site pond and wetlands area associated with the south diversion ditch.

Test sediment samples were collected from 8 sampling stations in the site wetlands area, 2 locations in the site pond, and 2 reference locations (Figure 3-1 and 3-2). For each sampling station, a 28-day subchronic survival and growth toxicity test using the amphipod *Hyalella azteca* was conducted in accord with standard protocols. The test results are summarized in Table 5-9.

As seen, statistically significant decreases in survival were noted for organisms exposed to sediments from 5 of 8 stations in the site wetlands area. Exposure to sediments collected from three stations (SD-2, SD-4, and SD-6) located in the northern portion of the wetlands resulted in

100% mortality. Statistically significant decreases in growth were observed for all sediments collected from the site wetlands area. No significant decreases in survival or growth were seen for sediment samples collected from the site pond.

These findings strongly support the conclusion that sediments in the wetlands area are likely to be causing adverse effects on populations of benthic receptors that may reside there. The data do not provide information on which chemicals are most likely to be responsible for the effects, or what the main source of the sediment contamination may be. However, HQ calculations based on measured sediment porewater concentrations from stations with decreased survival suggest that elevated levels of arsenic and zinc, and to a lesser extent antimony, cadmium, and lead, may account for the observed toxicity (see Section 5.2.2.4).

5.4 Tissue Burden Evaluation

Fish and aquatic invertebrates are exposed via multiple pathways, including direct contact with chemicals in surface water and sediment as well as ingestion of chemicals in sediment and dietary items. These exposures result in accumulation of chemicals in tissues, and the levels accumulated in the tissue are a direct measure of the total exposure from all routes. Table 5-10 presents the measured tissue burdens of metals in fish, benthic invertebrates, and snails collected from the wetlands area, the site pond, and reference locations. Interpretation of these results is limited because only one or two composite samples were collected from each area. However, as seen in Table 5-10, concentrations of several metals were higher in tissues from on-site organisms than organisms from reference locations, indicating that benthic invertebrates and snails at the RFT Site likely have increased exposure. No comparison was made for fish tissue because reference data were not available.

It is important to understand that increased exposure does not necessarily signify increased risk. Tissue burdens can be used as an indicator of the potential for toxic effects when compared to a tissue-based effects threshold. Jarvinen et al. (1999) provides a compilation of studies which provide tissue residues for several inorganic chemicals in aquatic receptors and the occurrence of adverse effects associated with the tissue burden. Appendix G provides detailed summaries of these studies and tissue levels associated with toxicity for all inorganics.

Table 5-10 provides the range of tissue concentrations associated with adverse impacts on growth, reproduction, or mortality across multiple species. As seen, measured concentrations of aluminum, lead, and zinc in fish from the site pond, and zinc in benthic invertebrates and snails from the wetlands area were above the tissue burden levels associated with the occurrence of adverse effects. These results suggest that aquatic organisms that reside in the wetlands area and site pond are exposed to levels of several metals that may have adverse effects on their survival and/or ability to grow and reproduce.

5.5 Aquatic Community Surveys

As described in the SLERA, only limited data exist on the fish community in Silver Creek near the site, and these data are all historic.

- A survey conducted in 1954 found a small number of trout (ATSDR, 1994)
- In 1970, fish were not present during electroshocking sampling (ATSDR, 1994)
- A 1986 investigation produced no fish (ATSDR, 1994)
- In 1991, cutthroat troat were reported to be present, but information regarding number of individuals or sampling locations is not available (E&E, 1991)
- Pan-sized trout were reportedly seen near the RFT Site in the spring of 1992 (USEPA, 1993a)

No recent data were located on the density or diversity of fish or benthic communities in Silver Creek or at the site.

5.6 Weight of Evidence Evaluation for Aquatic Receptors

As described in Section 4, the best approach for deriving reliable conclusions regarding risk is to combine the findings across all of the evaluation methods for which data are available, taking the relative strengths and weaknesses of each method into account. This approach is referred to as a weight-of-evidence evaluation. The individual lines of evidence which will be evaluated for aquatic receptors at the RFT Site are summarized in the following text table:

Exposure Pathway	Line of Evidence	Findings
Direct Contact with Surface Water	Estimated HQs from measured surface water concentrations	Surface water concentrations of cadmium and zinc in Silver Creek are probably adversely impacting aquatic receptors. Zinc may also be of concern to aquatic receptors in the site diversion ditch and wetlands area. Concentrations of several metals may be above a chronic level of concern in the unnamed drainage which flows into the site diversion ditch.
Direct Contact with Sediment	Estimated HQs from measured bulk sediment concentrations	Wide-spread, and potentially severe, toxicity to benthic invertebrates may be occurring in Silver Creek, the site diversion ditch, the wetlands area, and the site pond due to multiple metals in bulk sediment.
	Estimated HQs from measured sediment porewater concentrations	Sediment porewater concentrations of arsenic and zinc (antimony, cadmium and lead to a lesser extent) in the wetlands area, especially in the northern portion of the wetlands, may be of concern to benthic invertebrates.
	Sediment toxicity tests (*Hyalella azteca*)	Statistically significant deceases in survival were seen for 5 of 8 stations in the wetlands area. 100% mortality was seen in 3 sampling stations located in the northern part of the wetlands area.
All exposure pathways combined	Tissue burden evaluation	Measured tissue levels of zinc suggest that benthic invertebrates and snails in the wetlands area may be adversely impacted due to site exposures. Fish in the site pond may also be adversely impacted based on the elevated tissue levels of aluminum, lead, and zinc.
	Aquatic community evaluation	No recent data are available.

Based on these lines of evidence, it is concluded that metals in the wetlands area and the site diversion ditch are probably having an adverse effect on aquatic receptors (fish and aquatic invertebrates). Those metals which are likely to be the main risk drivers are antimony, arsenic, cadmium, lead, and zinc.

For Silver Creek, it is concluded that dissolved metals in surface water (especially cadmium and zinc) are likely to pose a significant risk to aquatic receptors. Because risks are elevated in surface water collected upstream of the RFT site, it is evident that sources besides the RFT site contribute to the toxicity. The headwaters of Silver Creek originate in the mountains south of Park City, a location that is influenced by several historic mining operations such as the Little Bell and Daly Mines. According to the findings of the Upper Silver Creek watershed evaluation (USEPA, 2001a), the Silver Maple Claims (Pace-Homer Ditch) was the largest contributor of zinc for the lower reaches of Silver Creek. Zinc loads from the RFT Site south diversion ditch are reported to contribute only 0.03 lbs/day to Silver Creek (USEPA, 2001a). Based on this information, it appears that the RFT Site is currently only a minor contributor to the current level of metal contamination in Silver Creek. However, if the metals present in sediments and/or surface water are reduced in Silver Creek as a result of off-site clean up activities, it may be possible that discharges from the RFT Site could recontaminate these media and become a more dominant influence on metal loading in the future.

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6 EVALUATION OF RISKS TO AMPHIBIANS

As discussed in Section 4, amphibians may be exposed to site contaminants via several potential exposure pathways. However, reliable exposure and toxicity data were only available to quantitatively evaluate exposures from direct contact with surface water.

Amphibians may be exposed to surface water at a number of locations including Silver Creek, the south diversion ditch, the wetlands area, the site pond, and an unnamed drainage which flows into the south diversion ditch (Figure 2-1). Evaluation of potential risks by the Hazard Quotient (HQ) approach, site-specific toxicity testing, and population surveys are summarized below.

6.1 HQ Approach: Direct Contact with Surface Water

HQ values for direct contact of amphibians with surface water are based on the ratio of the measured surface water concentration to an appropriate water toxicity benchmark, as follows:

$$HQ = \text{Exposure Conc}_{sw} / \text{Benchmark}_{sw}$$

where:

Conc_{sw} = chemical concentration in surface water (ug/L)
Benchmark_{sw} = chemical toxicity benchmark for water (ug/L)

The following sections describe the inputs used to calculate HQs and summarize the predicted risks to amphibians from surface water.

6.1.1 Exposure Assessment

Because concentrations of chemicals in surface water can vary significantly over time and location, exposure of aquatic receptors is best characterized as a distribution of individual values at each sampling location, rather than as an average of values over time and/or over location. Therefore, HQs were calculated for each sample for each chemical. In accord with USEPA guidance, non-detects were evaluated at one-half the detection limit.

For inorganics, concentration values in surface water may be expressed either as total recoverable or as "dissolved" (that which passes through a fine-pore filter). As noted above, there is general consensus that surface water toxicity for fish and benthic invertebrates is dominated by the level of dissolved chemicals, since chemicals that are adsorbed onto particulate matter may be less toxic than the dissolved forms. This is also expected to be true for amphibians, so exposures to inorganics in surface water were evaluated using dissolved concentrations.

Appendix B provides summary statistics (detection frequency, average, standard deviation, minimum, maximum) for each analyte in surface water for each exposure area. Raw surface water data are provided electronically in the site database.

6.1.2 Toxicity Assessment

Screening-level toxicity benchmarks for the protection of amphibians from direct contact with surface water were identified using the USEPA AQUIRE database. In most cases, the toxicity values available were LC50 values (the test concentration lethal to 50% of the test population). To estimate a toxicity benchmark value for no adverse effects, the lowest LC50 from the database was divided by a factor of ten. Table 6-1 provides the selected toxicity benchmarks for amphibians. It should be noted that these benchmarks should be interpreted as screening-level values that do not account for site-specific factors which may either increase or reduce toxicity.

6.1.3 Selection of COPCs

Because there were so few toxicity benchmarks available for amphibians, risks were evaluated for all chemicals detected in surface water for which toxicity benchmarks were available.

6.1.4 Risk Characterization

Appendix H presents the detailed calculations of HQ values for each surface water sample, along with graphs which summarize the distributions of HQ values for samples collected at each exposure reach. Two examples are presented in Figure 6-1 (arsenic) and Figure 6-2 (cadmium). In each figure, HQs based on non-detects are shown as open-circles and HQs based on detects are shown as closed circles. Note that the results in these figures are plotted on a log-scale, so large differences between HQ values are somewhat compressed.

The HQ distribution graphs presented in Appendix H were evaluated using the semi-quantitative approach for interpretation of HQs described in Section 4.4.1 and illustrated graphically in Figure 4-2. Table 6-2 summarizes the conclusions drawn from the graphs, based on a consideration of the number of exceedences (HQ > 1), the magnitude of the exceedences, the number of data points, and a comparison of site values to reference areas. Inspection of this table suggests the following main finding:

- Arsenic, cadmium, copper and/or lead in surface water appear to pose moderate to high risk to amphibians that may reside in Silver Creek, the site diversion ditch, and the unnamed drainages. Risks appear to be low in the wetlands area and the site pond. As noted earlier, contaminant levels in Silver Creek appear to be similar upstream and downstream of the site, making it difficult to quantify the contribution of the site to the contamination in Silver Creek.

6.2 Site-Specific Toxicity Tests

No site-specific toxicity tests were available for the RFT Site which evaluate amphibian exposures to environmental media.

6.3 Amphibian Community Surveys

No information was available on the amphibian community at the RFT Site.

6.4 Weight of Evidence Evaluation for Amphibians

Only one line of evidence was available to evaluate risks to amphibians from COPCs in surface water. The findings from this line of evidence are summarized in the following text table:

Exposure Pathway	Line of Evidence	Findings
Direct Contact with Surface Water	Estimated HQs from measured surface water concentrations	Surface water concentrations of arsenic, copper, and lead may be of concern to amphibians in Silver Creek and in RFT Site waters. Cadmium in surface water may also adversely impact amphibians in Silver Creek.

Based on this line of evidence, direct contact with metals in surface water may be having an adverse effect on amphibians in Silver Creek, the site diversion ditch and drainages, the wetlands area, and the site pond. The primary drivers for predicted risks are arsenic, cadmium, copper, and lead.

7 EVALUATION OF RISKS TO WILDLIFE RECEPTORS

As discussed previously in Section 4, the SLERA evaluated risks to terrestrial and aquatic/semi-aquatic wildlife and concluded that ingestion exposures from most media were potentially above a level of concern. Because no new data are available for contaminant levels in soils or terrestrial food web items, and because it is expected remedial activities will address concerns over soil-related pathways, this Baseline ERA does not re-evaluate terrestrial (upland) wildlife exposures. However, because new data are available for surface water, sediment, and aquatic food web items, exposures of aquatic/semi-aquatic wildlife from these pathways were quantitatively evaluated as described below.

7.1 HQ Approach: Ingestion Exposure Pathways

The basic equation used for calculation of an HQ value for exposure of a wildlife receptor to a chemical by ingestion of an environmental medium is:

$$HQ_{i,j,r} = \frac{Conc_{i,j} \times (IR_{j,r} / BW_r) \times DF_{j,r}}{TRV_{i,r}}$$

where:

$HQ_{i,j,r}$	=	HQ for exposure of receptor "r" to chemical "i" in medium "j"
$C_{i,j}$	=	Concentration of chemical "i" in medium "j" (e.g., mg/kg wet weight)
$IR_{j,r}$	=	Intake rate of medium "j" by receptor "r" (e.g., kg wet weight/day)
BW_r	=	Body weight of receptor "r" (kg)
$DF_{j,r}$	=	Dietary fraction of medium "j" by receptor "r" derived from site
$TRV_{i,r}$	=	Oral toxicity reference value for chemical "i" in receptor "r" (mg/kg-d)

Because all wildlife receptors are exposed to more than one environmental medium, the total Hazard Index (HI) to a receptor from a specific chemical is calculated as the sum of HQs across all media:

$$HI_{i,r} = \sum HQ_{i,j,r}$$

7.1.1 Selection of Representative Wildlife Species

It is not feasible to evaluate exposures and risks for each aquatic/semi-aquatic avian and mammalian species potentially present at the RFT Site. For this reason, several species were selected to serve as representative species (surrogates) of several different semi-aquatic feeding guilds. Selection criteria for representative wildlife species include trophic level, feeding habits, and the availability of life history information. Representative wildlife receptors selected for the RFT Site include:

Feeding Guild	Representative Species	Exposure Pathways Evaluated
Mammalian piscivore	Mink	Ingestion of surface water, sediment, and fish
Avian piscivore	Belted Kingfisher	
Avian omnivore	Mallard Duck	Ingestion of surface water, sediment, aquatic invertebrates, and aquatic plants
Avian insectivore	Cliff Swallow	Ingestion of surface water, sediment, and emerging aquatic insects

7.1.2 Exposure Assessment

Wildlife Exposure Factors

Exposure parameters and dietary intake factors for each surrogate wildlife receptor were derived from the Wildlife Exposure Factors Handbook (USEPA, 1993b), as well as a variety of other sources. The exposure parameters selected for each wildlife receptor are detailed in Appendix I and summarized in Table 7-1. Wildlife exposure factors were selected to represent average year-round adult exposures. When possible, exposure information was limited to receptor data from Utah or a representative western state. In some cases, no quantitative data could be located, so professional judgement was used in selecting exposure parameters. The dietary fraction (df) estimates were based on the average across all seasons.

Because the RFT Site is located in an area that is semi-arid, it is expected that wildlife (even those with larger home ranges) will be drawn to areas such as the site wetlands and pond to obtain aquatic prey and drinking water. Therefore, it was assumed that 100% of the total dietary intake of the representative receptor came from within the RFT Site.

Exposure Point Concentrations

Exposure areas for the evaluation of wildlife receptors were defined previously (see Section 3.3). Because wildlife receptors are generally mobile, exposure in an exposure area is related to the average concentration in each medium in the exposure area rather than the distribution of individual values within the area. However, because the true arithmetic mean concentration for an exposure area cannot be calculated with certainty from a limited number of measurements, the USEPA recommends that the upper 95th percentile confidence limit (UCL) of the arithmetic mean of the chemical concentrations be used to estimate exposure (USEPA, 1992). If the 95% UCL exceeds the highest detected concentration, then the highest detected concentration is used instead (USEPA, 1989). The resulting value (the 95% UCL or the maximum, whichever is lower) is referred to as the Exposure Point Concentration (EPC). When calculating an EPC, concentrations below the detection limit were evaluated by assuming a concentration value equal to one-half of the detection limit (USEPA, 1989).

The approach that is most appropriate for computing the 95% UCL of a data set depends on a number of factors, including the number of data points available, the shape of the distribution of the concentrations, and the degree of censoring (USEPA, 2002b). At the RFT Site, a simplified and conservative approach was used for estimating the UCL at an exposure area. Because most environmental data sets are found to be right skewed and are often well-approximated by a lognormal distribution, all UCLs were calculated using this approach.

Table 7-2 provides a summary of the EPC values in surface water, sediment, and food items (fish, benthic invertebrates/snails, wetland vegetation) used to evaluate ingestion exposures for wildlife in the Baseline ERA. For emergent insects, no data were collected. Therefore, concentrations in emergent insect tissues were estimated to be equal to concentrations measured in benthic invertebrates.

7.1.3 Toxicity Assessment

Selection of Toxicity Reference Values

A Toxicity Reference Value (TRV) for wildlife provides an estimate of the dose (in units of mg of chemical per kg of body weight per day, mg/kg/day) associated with a known effect. Often, two types of dose-based TRVs are identified. The first TRV is an estimate of the dose that is not associated with any adverse effects, and is referred to as the no observed adverse effect level (NOAEL) TRV. The second TRV is an estimation of the dose that causes an observable adverse effect, and is referred to as the lowest observed adverse effect level (LOAEL) TRV. This range of TRVs is one way to bracket the true threshold for adverse effects.

It is expected that the adverse effect threshold will vary from species to species within any particular taxonomic group. If data are available of the effects thresholds for many different species in a particular group, the data may be rank-ordered to define a species-sensitivity distribution (SSD) for that group. In order to ensure that the HQs calculated for each representative species are protective of most species within the group, a TRV which represents the lower end of the SSD is preferred. Ideally, toxicity data would be sufficient to define the SSD and support derivation of a TRV for each unique feeding guild selected for evaluation (e.g., avian omnivores, mammalian herbivores, etc.). Unfortunately, available toxicity data for birds and mammals are generally not robust enough to develop SSDs for each feeding guild, so a single bird TRV and mammal TRV were used to represent all bird and mammal species, respectively.

Because the purpose of the Baseline ERA is to evaluate wildlife exposures from ingestion of contaminated media at the RFT Site over the lifetime of the receptor, TRVs were derived from studies in which the exposure route was oral (eg: via ingestion in diet or water or via gavage), and dosing occurred over a long period of time (chronic exposure) or during a critical lifestage period. The wildlife TRVs were selected to represent relevant toxicity endpoints for population sustainability (eg: growth, reproduction, mortality).

TRVs for wildlife were compiled from three secondary sources (shown in order of preference): USEPA (2003b), Engineering Field Activity West (1998), and Sample et al. (1996). Appendix C provides a summary of the TRV derivation approach and the bird and mammal TRVs selected by each secondary source. The TRVs provided in each of these sources are described briefly below.

In USEPA (2003b), a single bird TRV and mammal TRV was derived which represents the highest no effect level below the level which effects are first observed across multiple species and endpoints. Risk calculations in the Baseline ERA used this TRV without adjustment.

In Engineering Field Activity West (1998) and Sample et al. (1996), two types of TRV are provided for both birds and mammals. Risk calculations in the Baseline ERA were based on the geometric mean of the selected NOAEL (or Low TRV in Engineering Field Activity West, 1998) and LOAEL (or High TRV). This geomean value was used as an estimate of the threshold dose level where adverse effects first begin to occur in exposed organisms. If only a NOAEL was available, this value was used to represent the effects threshold.

Table 7-3 summarizes the mammal and bird TRVs that were used to evaluate potential risks to representative wildlife species.

Adjustments for Relative Bioavailability

TRVs from literature studies are generally expressed in units of ingested dose (mg of chemical per kg of body weight per day, mg/kg/day). However, the toxicity of an ingested dose depends on how much of the ingested dose is actually absorbed, which in turn depends on the properties of both the chemical and the exposure medium. Ideally, toxicity studies would be available that establish empiric TRVs for all site media of concern (water, food, soil, sediment). However, most laboratory tests use either food or water as the exposure medium, and essentially no studies use soil or sediment. Therefore, in cases where a TRV is based on a study in which the oral absorption fraction is different that what would be expected for a site medium, it is desirable to adjust the TRV to account for the difference in absorption whenever data permit.

The ratio of absorption from the study medium compared to absorption from site medium is referred to as the relative bioavailability (RBA). The RBA is used to adjust the TRV as follows:

$$TRV(adjusted) = TRV(literature) / RBA$$

For the purposes of this assessment, it was assumed that chemicals are absorbed equally well from all site media (water, diet, sediment) and the RBA was equal to 1.0 (100%). This approach is likely to be realistic for contaminants in water and most food web items, but may tend to overestimate exposure and risk from ingestion of sediment. However, no site-specific information on RBA was available which would provide a basis to modify the RBA from sediment.

7.1.4 Selection of COPCs

Based on the COPC selection results from the SLERA, only a few chemicals could be excluded as COPCs for exposure of wildlife. Therefore, a selection of COPCs was not performed in the Baseline ERA and risks were evaluated for all chemicals detected in abiotic and biotic media which were not essential nutrients [3].

7.1.5 Risk Characterization

Based on the results of the SLERA (USEPA, 2003a), risks were predicted to be above a level of concern for aquatic/semi-aquatic wildlife from ingestion of sediment and aquatic prey items in Silver Creek. No new sediment or aquatic food data were collected for Silver Creek since the SLERA. Therefore, predicted risks from these pathways have not changed since the SLERA and were not re-evaluated in the Baseline ERA.

Tables 7-4 to 7-7 provide the detailed risk calculations for each wildlife receptor for each chemical of potential concern at each exposure area. Each table (one table per receptor) shows the predicted HQ and HI values for each chemical in each site-related exposure area. In addition, predicted HQs and HIs are also shown for two reference areas (one wetland, one pond) for most media (fish tissue data were not collected from either reference location). A comparison of predicted site risks relative to predicted risks for reference areas helps to identify cases where predicted risks are above a level of concern not only at the site but also at the reference area. In this case, it is possible that the exposure and/or the toxicity assumptions for the chemical are overly conservative, since risks are not expected to be of concern in non-impacted reference areas.

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[3] Essential nutrients include: calcium, magnesium, iron, potassium, and sodium (USEPA, 1997).

Table 7-8 presents the primary contaminants (relative to reference) and exposure pathways that are predicted to contribute the most risk for each wildlife receptor.

- For the mink (Table 7-4), estimated risks were below a level of concern for all chemicals at all exposure areas.

- For the mallard duck, belted kingfisher, and cliff swallow (Tables 7-5 to 7-7), the primary contributor to estimated risks was incidental ingestion of lead in sediments from the wetlands area, the south diversion ditch, and site pond. For the cliff swallow (Table 7-7), estimated risks were above a level of concern for manganese and zinc at the site wetlands area and the site pond from ingestion of aquatic invertebrates and sediment. However, because the measured aquatic invertebrate concentrations of manganese did not correlate well with measured sediment concentrations of manganese, and because sediment concentrations in some reference locations were higher than site, it is not certain whether the predicted risk from manganese is of authentic concern.

- Risks to wildlife from surface water ingestion were below a level of concern for all chemicals for all receptors.

7.1.6 Species-Specific Interpretation of Risk Estimates

It is important to remember that the HQ and HI values presented above are based on TRV values that take inter-species variability in sensitivity into account and are intended to be protective of nearly all species within the feeding guild evaluated. Because of this, when the calculated HQ or HI for a feeding guild is found to exceed 1, it is not necessarily true that all species comprising the guild will be at risk. Rather, an HQ or HI above 1 implies that the most sensitive species in the guild are likely to be at risk, and risk may or may not extend to other less sensitive species in the guild. Thus, in some cases it may be informative to estimate risks for selected species within a guild in order to better understand the impacts on the different species within the guild.

For example, using the TRV for lead based on all bird data (1.6 mg Pb/kg-BW/day) (USEPA 2003b), an HI value of 16 is predicted for exposure of waterfowl to lead in the wetlands area. If only lead toxicity data for the mallard are considered, the TRV is about 20 mg Pb/kg-BW/day (more than 10 times higher than the TRV used to represent all bird species). Hence, the predicted HI value for the mallard (1.4) is about 10-times lower than predicted from the all-bird TRV. This indicates that mallards are apparently not among the more sensitive species of birds to lead toxicity, but it does not imply that risks to other species of waterfowl or other species of birds that may be exposed are also overestimated. Rather, it is correct to conclude, as described above, that some species of birds in the guild are likely to be at risk from lead, even though mallards may not be among the threatened species.

7.2 Site-Specific Toxicity Tests

No site-specific toxicity tests were available which evaluate wildlife exposures to environmental media from the RFT Site.

7.3 Wildlife Community Surveys

Wildlife population surveys and community evaluations have not been conducted at the RFT Site.

7.4 Weight of Evidence Evaluation for Wildlife Receptors

Only one line of evidence (the HQ/HI approach) was available to evaluate risks to wildlife receptors from COPCs in surface water, sediment, and the diet. The findings from this line of evidence are summarized in the following text table:

Exposure Pathway	Line of Evidence	Findings
Ingestion of surface water, sediment, and aquatic food items	Estimated HQs and HIs from ingested dose (calculated from measured data)	Risks to birds are likely to be of potential concern in the wetlands, diversion ditch, and pond, primarily from lead in sediment and also from these lead in aquatic food items. Risks to mink were below a level of concern for all chemicals at all exposure areas. Risks to the cliff swallow may be above a level of concern from manganese and zinc in aquatic invertebrates and sediment. However, correlation of manganese in sediment compared to manganese in invertebrates is inconsistent, so predicted risks may not be site-related or may reflect an overly conservative TRV.

Based on this line of evidence, it was concluded that incidental ingestion of lead, manganese and zinc in sediments from the wetlands area, the south diversion ditch, and site pond are likely to be causing adverse effects in waterfowl and other birds which feed in these areas. Concentrations of lead, and possibly zinc and manganese, in aquatic food items may also cause adverse effects in birds that consume fish, aquatic invertebrates, or aquatic plants from the RFT Site.

8 UNCERTAINTIES

Quantitative evaluation of ecological risks is generally limited by uncertainty regarding a number of important data. This lack of knowledge is usually circumvented by making estimates based on whatever limited data are available, or by making assumptions based on professional judgement when no reliable data are available. Because of these assumptions and estimates, the results of the risk calculations are themselves uncertain, and it is important for risk managers and the public to keep this in mind when interpreting the results of a risk assessment.

The following text summarizes the key sources of uncertainty influencing the results of this Baseline ERA.

8.1 Uncertainties in Nature and Extent of Contamination

Representativeness of Samples Collected

Concentration levels of chemicals in environmental media are often quite variable as a function of location, and may also vary significantly as a function of time. Thus, samples collected during a field sampling program may or may not fully characterize the spatial and temporal variability in actual concentration levels. At this site, field samples were collected in accord with sampling and analysis plans that specifically sought to ensure that samples were representative of the range of conditions across each exposure area. However, in some locations, the number of samples collected was relatively small. Thus, without the collection of very large numbers of samples over both space and time, some uncertainty remains as to whether the samples collected provide an accurate representation of the distribution of concentration values actually present.

Accuracy of Analytical Measurements

Laboratory analysis of environmental samples is subject to a number of technical difficulties, and values reported by the laboratory may not always be exactly correct. However, data used in this risk assessment had sufficient accompanying quality assurance data to ensure that results were within acceptable bounds for accuracy and precision. The magnitude of analytical error is usually small compared to other sources of uncertainty, although the relative uncertainty increases for results that are near the detection limit.

8.2 Uncertainties in Problem Formulation

Exposure Pathways Not Evaluated

Exposure pathways selected for quantitative evaluation in this Baseline ERA do not include all potential exposure pathways for all ecological receptors. Exposure pathways that were not evaluated include:

- Ingestion of prey items and sediments by benthic invertebrates
- Dermal exposures of wildlife to sediment and surface water
- Inhalation of dust particles by wildlife
- Ingestion and direct contact exposures in amphibians and reptiles

Omission of these pathways will tend to lead to an underestimation of total risk to the exposed receptors. However, as discussed previously, most of these exposure pathways are likely to be minor compared to other pathways that were evaluated, and the magnitude of the underestimation is not likely to be significant in most cases.

One possible exception is ingestion of prey items by benthic invertebrates and fish. Although the general consensus is that uptake of inorganic contaminants from food is usually less than from direct contact with water (Clements, 1991), available data are sufficient to indicate that the ingestion pathway can be an important source of exposure to some aquatic receptors (Timmermans et al., 1992), and that dietary exposures can be capable of limiting growth in at least some cases (Duddridge and Wainwright, 1980). Thus, omission of the ingestion pathway for aquatic receptors is likely to be a minor source of uncertainty in most cases, but could lead to an underestimate in some cases.

8.3 Uncertainties in Exposure Assessment

Chemicals Not Detected

Any chemical that was never detected in a site medium was not evaluated in exposures of receptors to that medium. However, in some cases, the analytical detection limit was too high to expect the chemical would have been detected even if it were present at the level of concern. Chemicals in this category were assigned to the Type 2 Qualitative COPC category. The COPC selection tables for each receptor class and media (Table 5-2, Table 5-5, and Table 5-7) identify chemicals assigned to this category. As seen, a number of such chemicals exist. Omission of these chemicals is likely to result in an underestimation of risk. However, it is suspected that the magnitude of the underestimation is likely to be low in most cases. This is because, if the non-detected chemical were actually site-related and were present at a level of substantial health concern, it likely would have occurred at levels above the detection limit at least a few times. Thus, while the hazard from Type 2 Qualitative COPCs is unknown, it is probably not large enough to cause a substantial underestimation of risk.

Exposure Area Concentration Values

In all exposure calculations, the desired input parameter is the true mean concentration of a chemical within a medium, averaged over the area where random exposure occurs. However, because the true mean cannot be calculated based on a limited set of measurements, the USEPA (1989, 1992) recommends that the exposure estimate be based on the 95% upper confidence limit of the mean. When data are plentiful and inter-sample variability is not large, the EPC may be only slightly higher than the mean of the data. However, when data are sparse or are highly variable, the EPC may be far greater than the mean of the available data. Such EPCs (substantially higher than the sample mean) reflect the substantial uncertainty that exists when data are sparse or highly variable, and in general are likely to result in an overestimate of risk.

Wildlife Exposure Factors

The intake (ingestion) rates for food, soil, water, and sediment used to estimate exposure of wildlife at the site are derived from literature reports of intake rates, body weights, dietary compositions, consumption rates, and metabolic rates in receptors at other locations or from measurements of laboratory-raised organisms. These values may or may not serve as appropriate models for site-specific intake rates of wild receptors at this site. Moreover, the actual dietary composition of an organism will vary daily and seasonally. In addition, some wildlife receptor-specific intake rates are estimated by extrapolation from data on a closely related species or by use of allometric scaling equations (scaling of intake rates based on body weights). This introduces further uncertainty into the exposure and risk estimates. These uncertainties could either under- or overestimate the actual exposures of wildlife to chemicals in water, sediment, and diet.

For this analysis, it was also assumed that wildlife exposures were continuous and that receptor home ranges were located entirely within the RFT Site (i.e., all of the total dietary intake was from the site). In the case of resident small-home range receptors, these assumptions are likely to be fairly realistic. However, these assumptions may tend to overestimate receptors that have large home range and that may not be exposed on-site most of the time.

Absorption From Ingested Doses

The toxicity of an ingested chemical depends on how much of the chemical is absorbed from the gastrointestinal tract into the body. However, the actual extent of chemical absorption from ingested media (soil, sediment, food, and water) is usually not known. The hazard from an ingested dose is estimated by comparing the dose to an ingested dose that is believed to be safe, based on tests in a laboratory setting. Thus, if the absorption is the same in the laboratory test and the exposure in the field, then the prediction of hazard will be accurate. However, if the absorption of chemical from the site medium is different (usually lower) than occurred in the laboratory study, then the hazard estimate will be incorrect (usually too high). In this assessment, estimates of wildlife exposure assumed a relative bioavailability (RBA) of 100% for all chemicals in all media. This assumption is expected to be reasonable for chemicals in surface water and most dietary food items, but may tend to overestimate exposure for exposure to chemicals in soil and sediment. This is because metals in soil and sediment may occur in mineral phases that have low solubility, and this tends to reduce the amount of metal that is absorbed when ingested.

8.4 Uncertainties in Toxicity Assessment

Representativeness of Receptors Evaluated

Risk characterizations for aquatic receptors are based on a generalized set of species found in freshwater aquatic communities. However, not all of these species (e.g.: fish) are expected to occur in waters at the RFT Site. Thus, HQ values above 1 may reflect risks to species that are absent at the site, and risks to species that are actually present at the site may be lower.

Risks to wildlife are assessed for a small subset of the species likely to be present at the RFT Site. Although the wildlife species selected quantitative evaluation at this site represent a range of taxonomic groups and life history types of species likely to occur in the area, these species may not represent the full range of sensitivities present. The species selected may be either more or less sensitive to chemical exposures than typical species located within the area.

Absence of Toxicity Data for Some Chemicals

For a number of chemicals that were detected in one or more samples of site media, no reliable toxicity benchmark could be located for one or more receptor types. Chemicals in this category were assigned to the Type 1 Qualitative COPC category. The COPC selection tables for each receptor class and media (Table 5-2, Table 5-5, and Table 5-7) identify chemicals assigned to this category. As seen, a number of such chemicals exist. The inability to evaluate hazard from these chemicals is expected to result in an underestimation of risk, but it is suspected that the magnitude of the error is usually likely to be low. This is because the absence of a toxicity benchmark for a chemical is most often because toxicological concern over that chemical is low. That is, chemicals that lack benchmarks are often considered to be relatively less hazardous that those for which benchmarks do exist. To the extent that this is true (even though there are likely some exceptions to this rule), risks from Type 1 Qualitative COPCs are likely not to contribute risks of the same magnitude as those predicted for chemicals that do have a benchmark value.

Extrapolation of Toxicity Data Between Receptors

Toxicity data are not available for all of the species of potential concern at the site. Thus, it is sometimes necessary to estimate toxicity values for a receptor by extrapolating toxicity data across similar species. At this site, this extrapolation was direct: that is, no uncertainty factor was used to adjust a benchmark form one species when applied to another. This approach may either overestimate or underestimate the risk to the actual receptor, depending on whether the actual receptor is less sensitive or more sensitive that the species for which data are available, and the magnitude of the error could be significant in some cases.

Extrapolation of Toxicity Data Across Dose or Duration

In some cases, TRV data are available only for high dose exposures, and extrapolation to low doses (similar to those that actually occur at the site) is a source of uncertainty. Likewise, some TRVs are based on relatively short-term exposures, and extrapolation to long-term exposures is uncertain, especially for chemicals that tend to build up in the exposed organism. When such extrapolations are necessary, it is customary to include one or more "uncertainty factors" in the derivation of the benchmark to account for the extrapolation. In general, these "uncertainty factors" are likely to be somewhat too large, so the benchmarks derived in this way are more likely to overestimate than underestimate true risk.

Extrapolation of Toxicity Data from Laboratory to Field Conditions

Even when data are available for a species of concern at the site, the data are usually generated under laboratory conditions and extrapolation of those data to free-living receptors in the field is uncertain. In some cases, site-specific factors may tend to modify (often decrease) the toxicity of chemicals in surface water, sediments, and soil. For example, metals in surface water may be bound to soluble organic materials that reduce the tendency for the metal to bind to respiratory structures of benthic organisms or fish. Similarly, the presence of organic matter in soil, along with other substances, may have a significant influence on actual toxicity to plants and soil organisms. Thus, risks based on literature-derived toxicity factors may sometimes overestimate risk from site media.

8.5 Uncertainties in Risk Characterization

Interactions Among Chemicals

Most toxicity benchmark values are derived from studies of the adverse effects of a single contaminant. However, exposures to ecological receptors usually involve multiple contaminants, raising the possibility that synergistic or antagonistic interactions might occur.

However, data are generally not adequate to permit any quantitative adjustment in toxicity values or risk calculations based on inter-chemical interactions. In accordance with USEPA guidance, effects from different chemicals are not added unless reliable data are available to indicate that the two (or more) chemicals act on the same target tissue by the same mode of action. At this site, HQ values for each chemical were not added across different chemicals. If any of the chemicals of concern at the site act by a similar mode of action, total risks could be higher than estimated.

Estimation of Population-Level Impacts

Assessment endpoints for the receptors at this site are based on the sustainability of exposed populations, and risks to some individuals in a population may be acceptable if the population is expected to remain healthy and stable. However, even if it is possible to accurately characterize the distribution of risks or effects across the members of the exposed population, estimating the impact of those effects on the population is generally difficult and uncertain. The relationship between adverse effects on individuals and effects on the population is complex, depending on the demographic and life history characteristics of the receptor being considered as well as the nature, magnitude and frequency of the chemical stresses and associated adverse effects. Thus, the actual risks that will lead to population-level adverse effects will vary from receptor to receptor. In this Baseline ERA, an assessment of the risk of population effects was based on professional judgement, considering both the frequency and the magnitude of HQ exceedences. These judgements are not certain and should be interpreted accordingly.

8.6 Summary of Uncertainties

Table 8-1 summarizes the various sources of uncertainty in this Baseline ERA, along with a qualitative estimate of the direction and magnitude of the likely errors attributable to the uncertainty. Based on all of these considerations, the HQ and HI values calculated and presented in this Baseline ERA should be viewed as having substantial uncertainty. Because of the inherent conservatism in the derivation of many of the exposure estimates and toxicity benchmarks, these HQ and HI values should generally be viewed as being more likely to be high than low, and results and conclusions should be interpreted accordingly.

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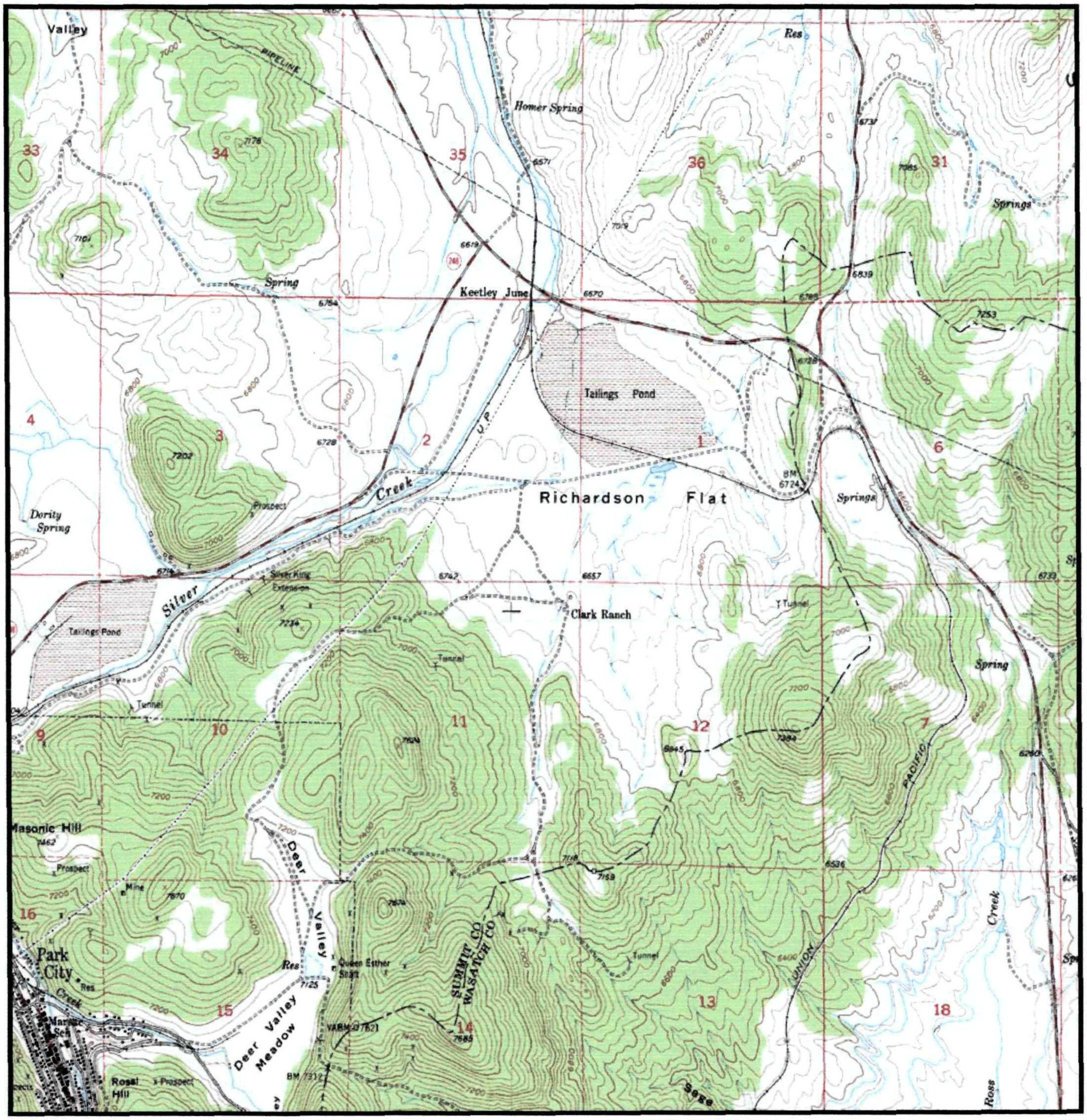


Figure 1 - 1
Richardson Flat Tailings Site Location Map

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

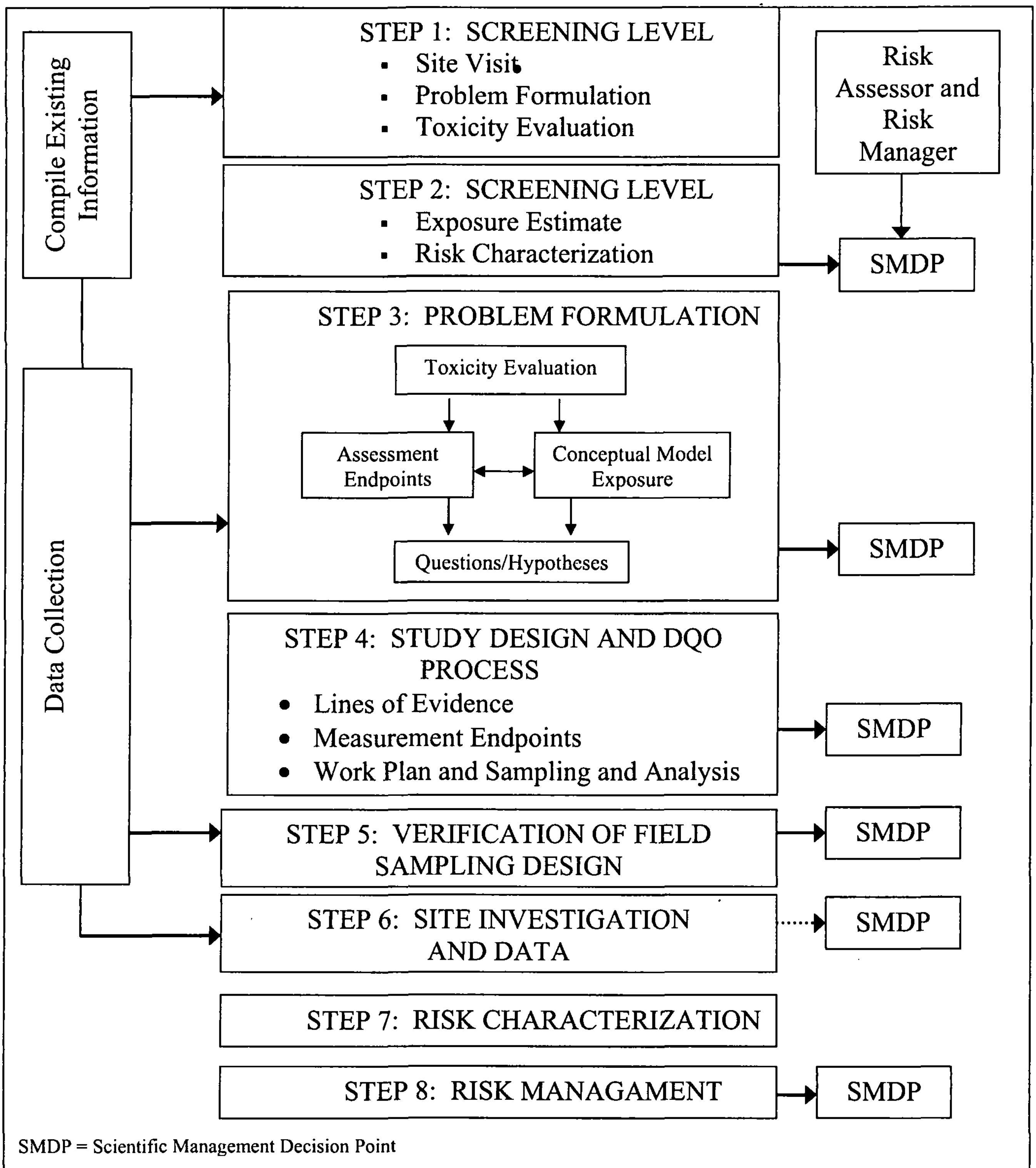


Figure 1-2
Eight Step Process Recommended in Ecological Risk Assessment Guidance for Superfund (ERAGs) (USEPA, 1997)

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

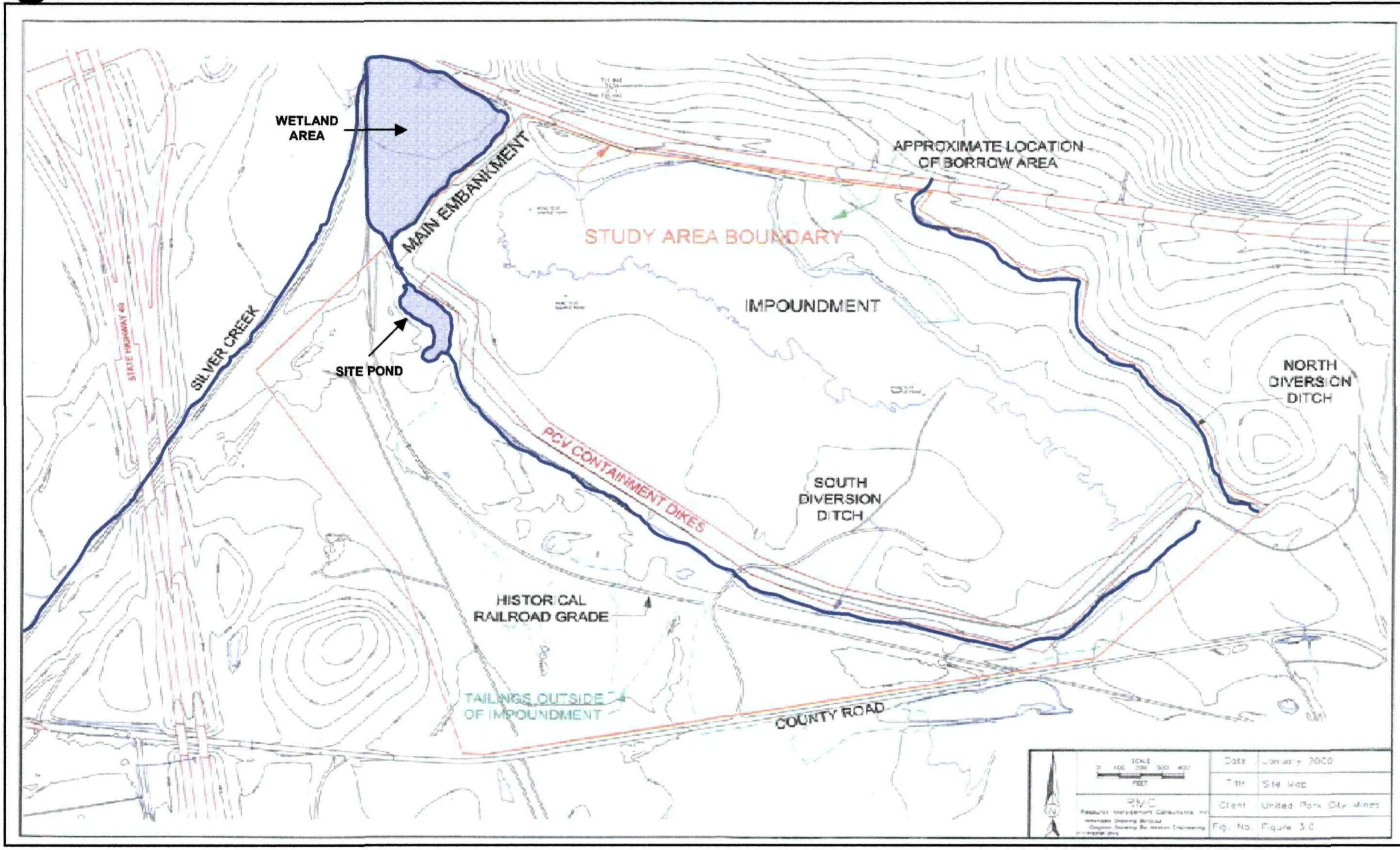


Figure 2-1
Richardson Flat Tailings Site Features

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Fig 2-1 RFT Site Map.ppt

Source: Resource Management Consultants, 2000 (RI/FS Workplan)

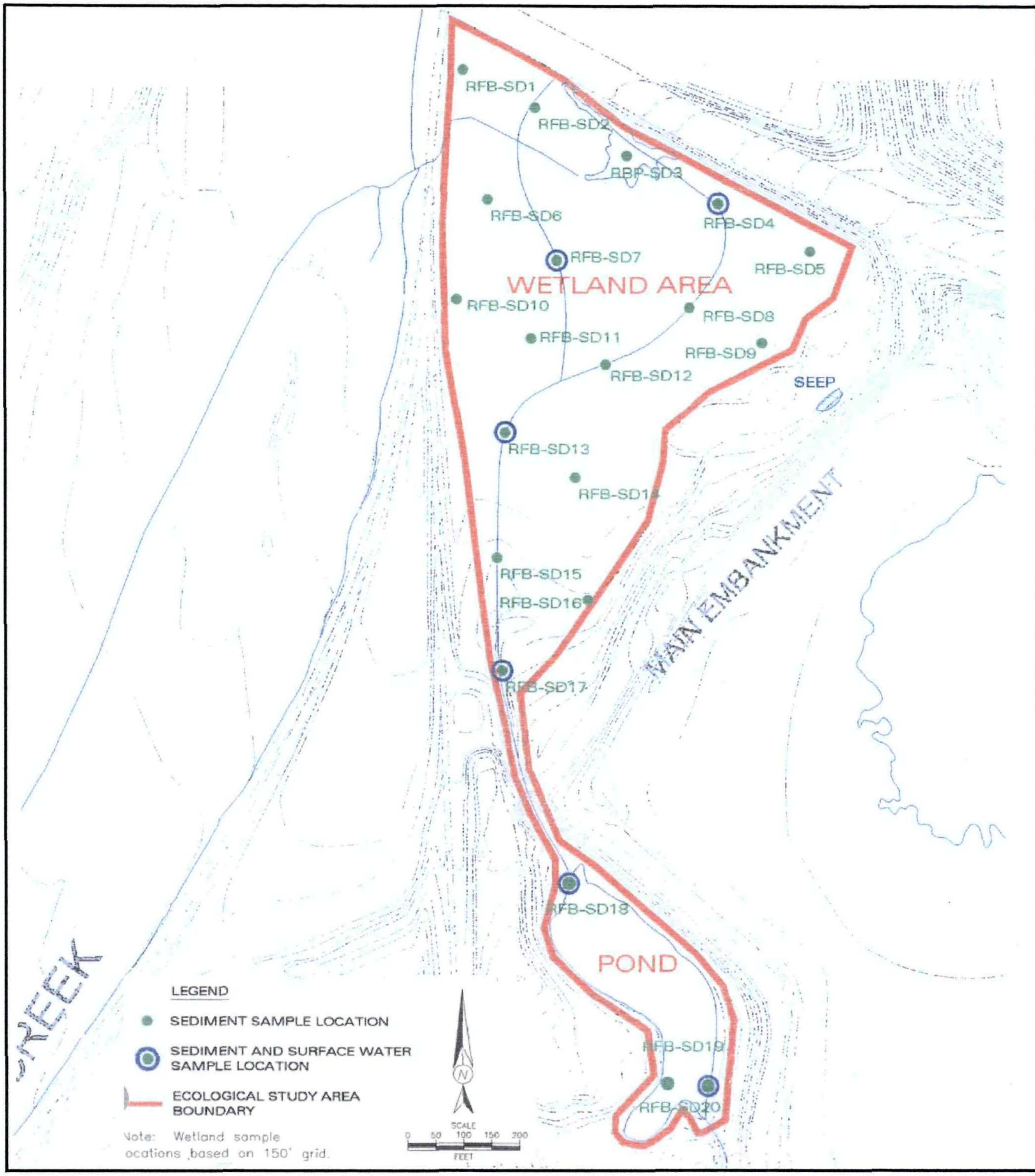


Figure 3-1
Locations in the Wetlands and Pond Sampled During the Phase I/II Field Investigations

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Source: Resource Management Consultants, 2003 (Wetland/Pond SAP)

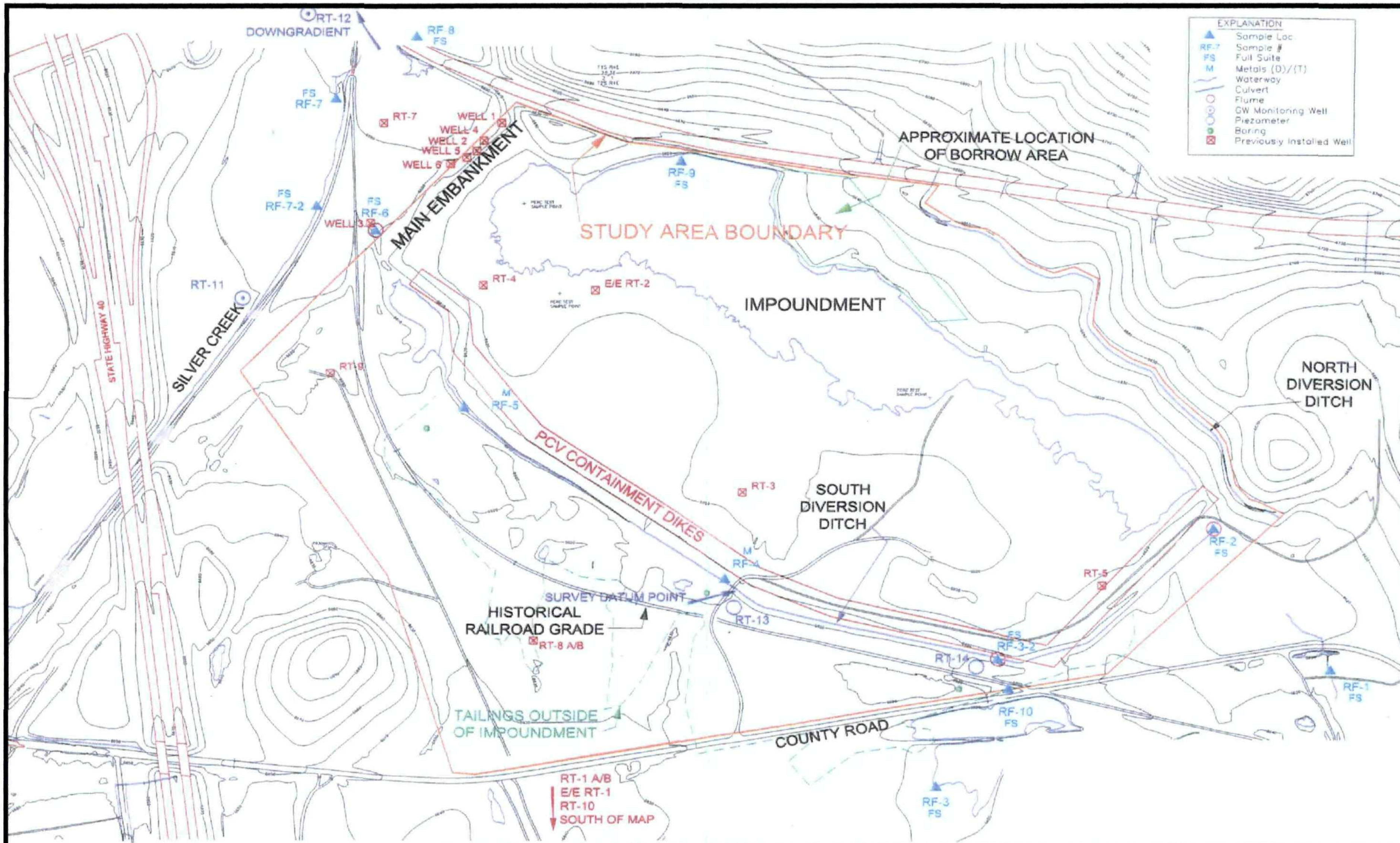
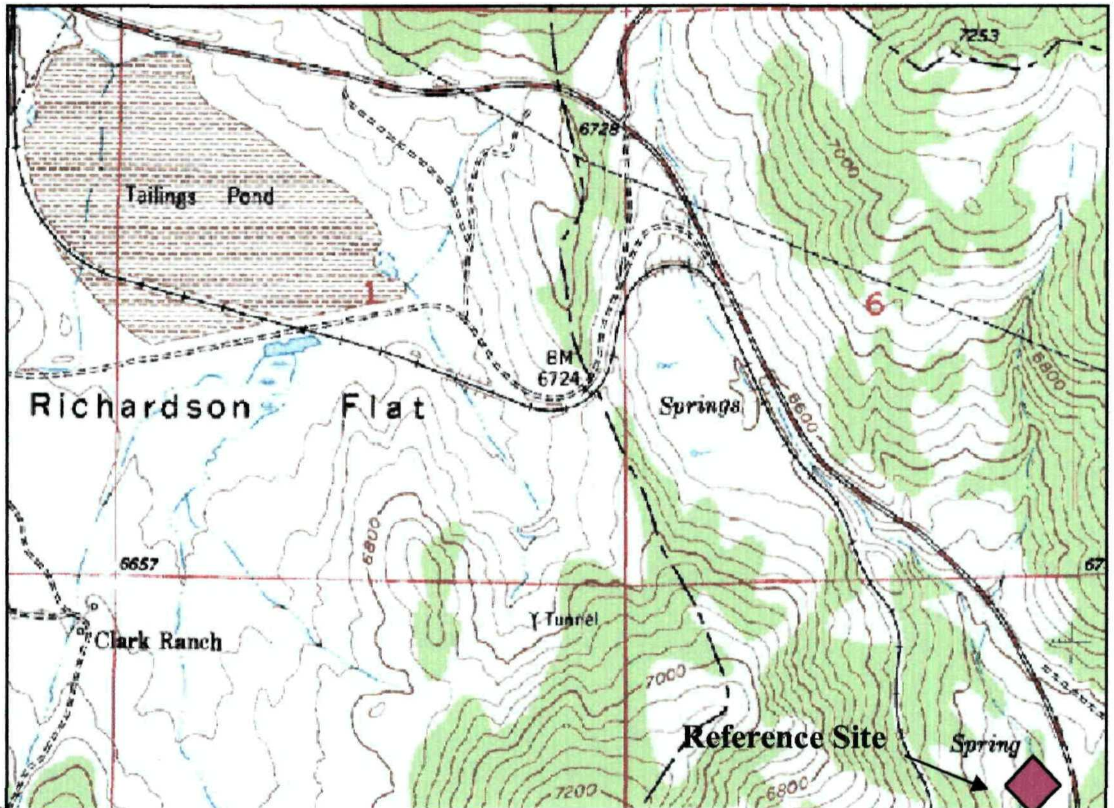


Figure 3-2
Surface Water and Groundwater Monitoring Locations

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Fig 3-2 RMC Locations.ppt

Source: Resource Management Consultants, 2000a [RI/FS Workplan]



Location Relative to Richardson Flat Tailings Site

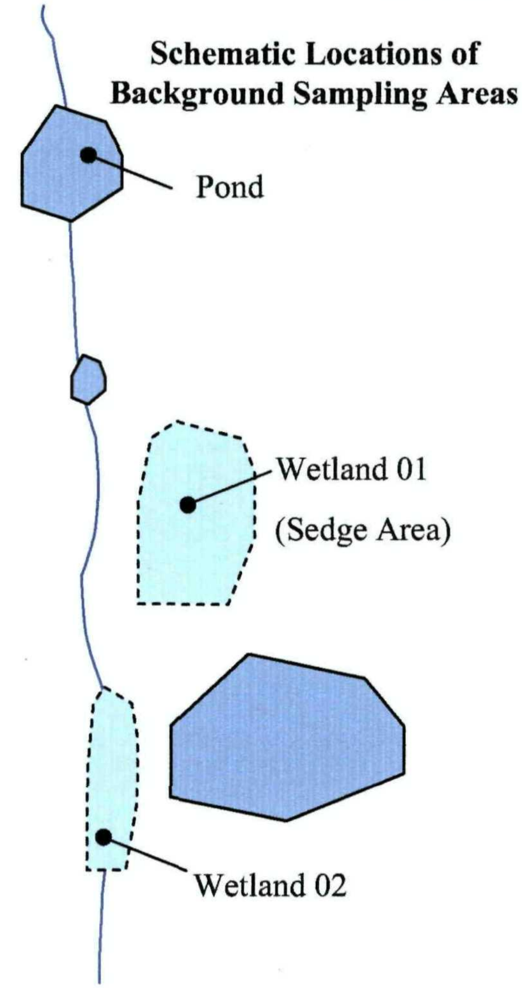


Figure 3-3
Background Sampling Locations for Aquatic Environments

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Source: Resource Management Consultants, 2003 – September Monthly Report

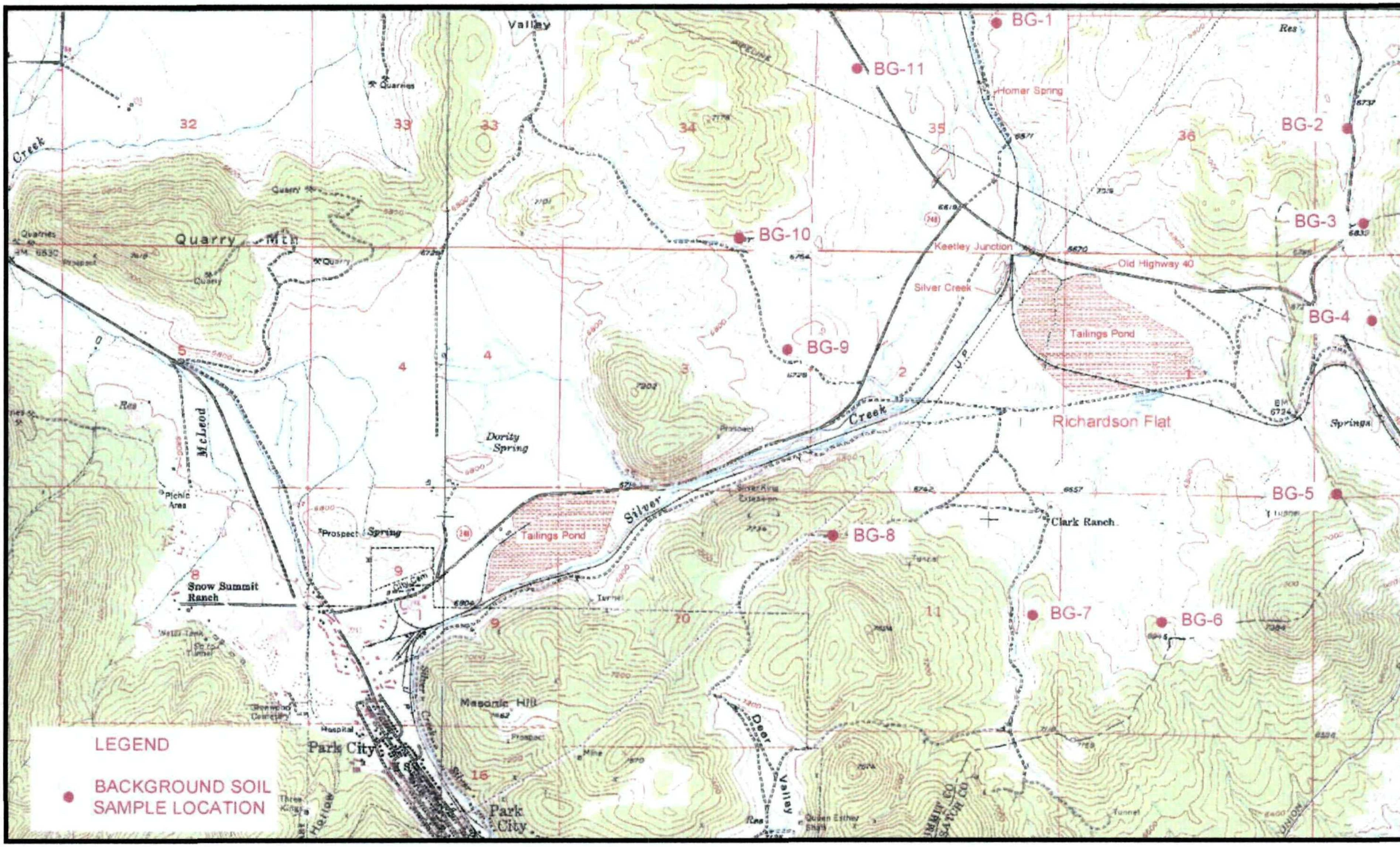


Figure 3-4
Background Sampling Locations for Terrestrial Environments

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Source: Resource Management Consultants, 2000a [RI/FS Workplan]

Figure 4-2
Conceptual Approach for Characterizing Population-Level Risks

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

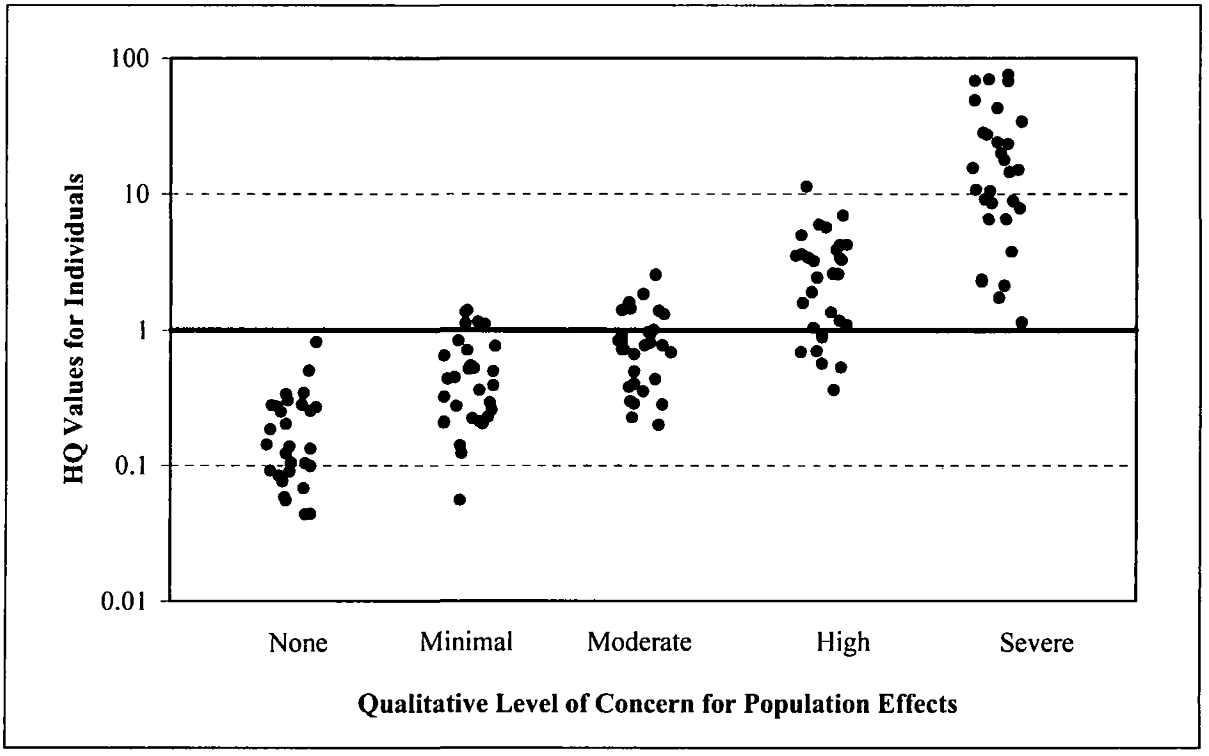
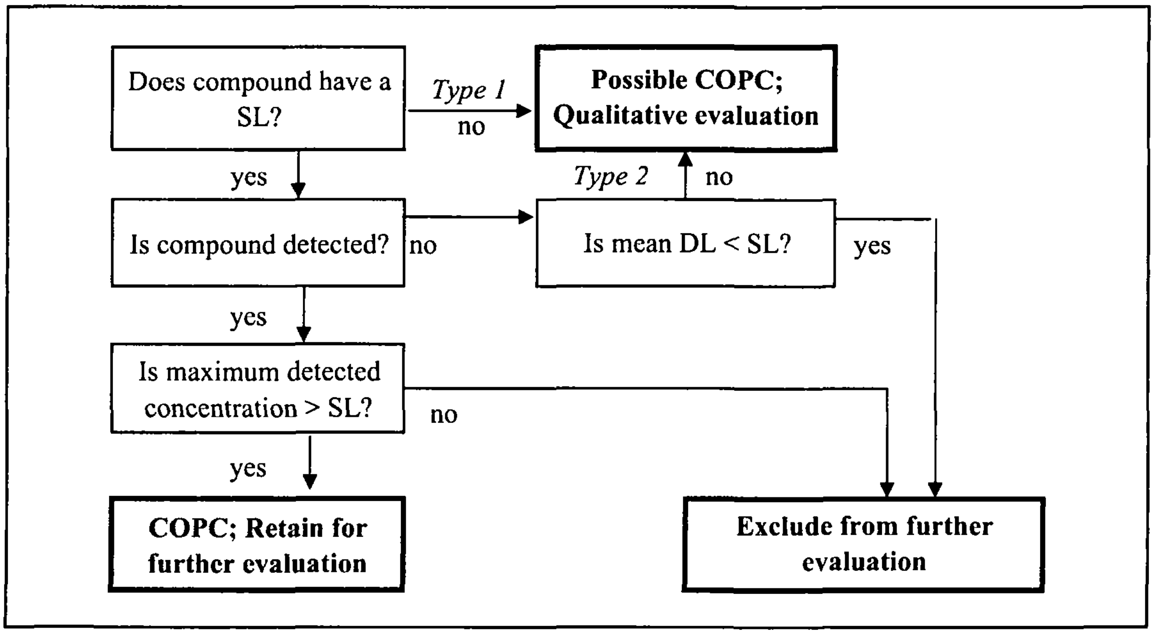


Fig 4-2 HQ Approach.xls

Figure 5-1
Procedure for Identifying Chemicals of Potential Concern (COPCs)

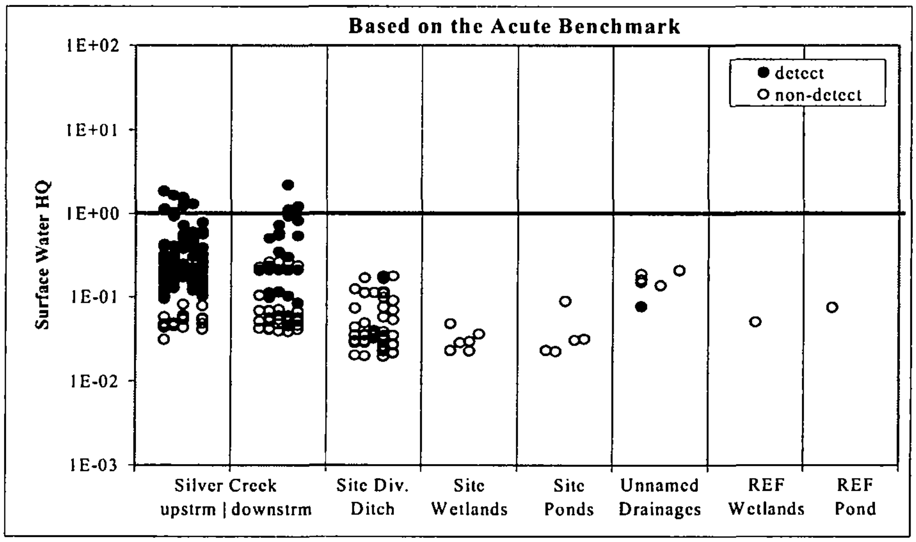
Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

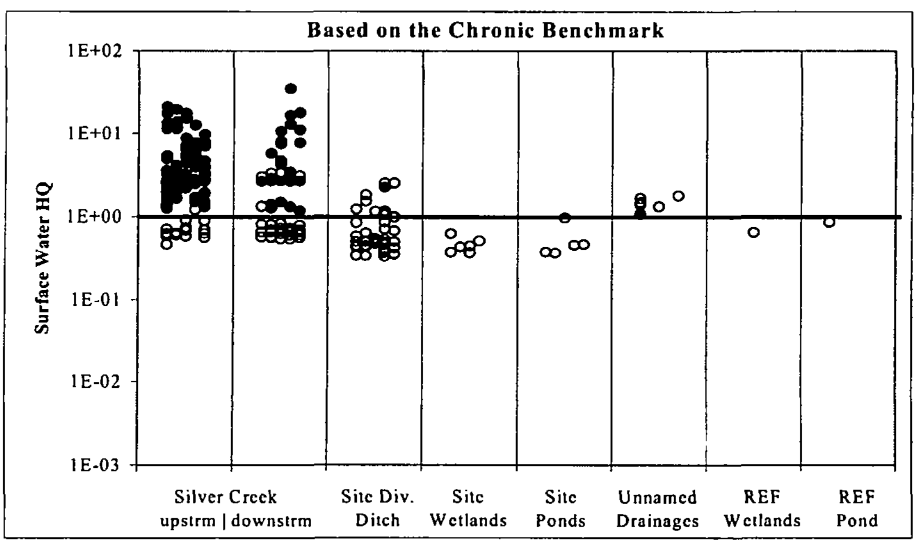


SL = Screening Level Benchmark
DL = Detection Limit

Figure 5-2
Evaluation of Risks to Aquatic Receptors from
Direct Contact with Dissolved Cadmium in Surface Water

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site



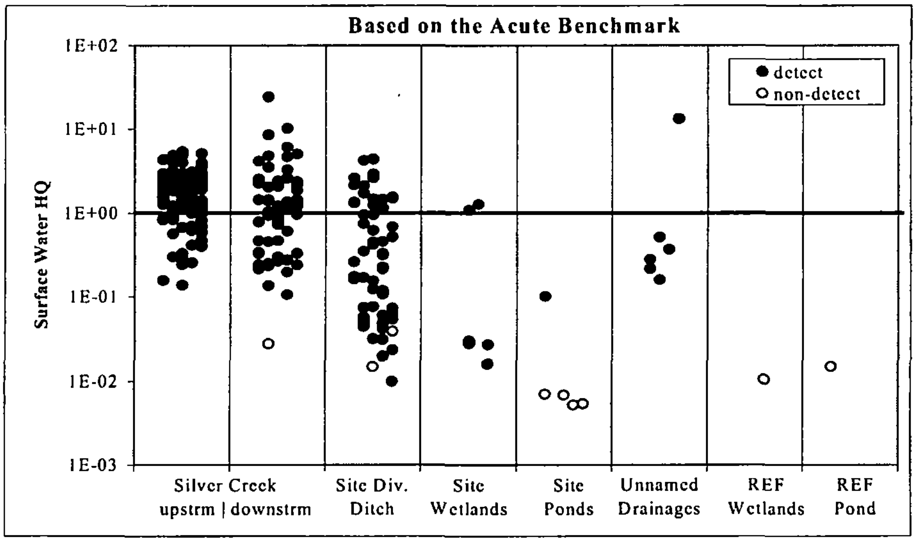


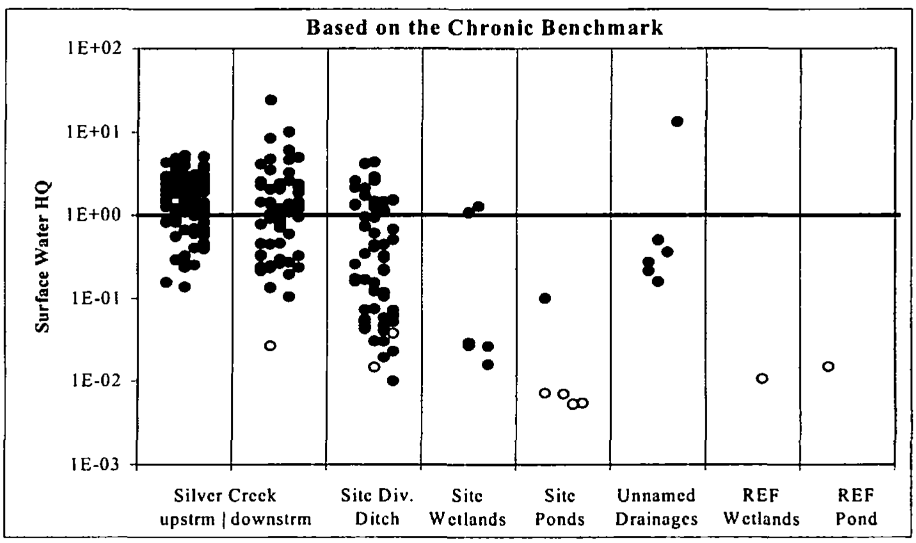
Location	N samples > benchmark			
	acute		chronic	
Silver Creek - upstream	12/109	*11%*	87/109	*80%*
Silver Creek - downstream	3/63	*5%*	31/63	*49%*
Site Diversion Ditch	0/59	*0%*	12/59	*20%*
Site Diversion Ditch - Wetlands Area	0/6	*0%*	0/6	*0%*
Site Pond	0/4	*0%*	0/4	*0%*
Unnamed Drainages	0/6	*0%*	6/6	*100%*
Reference Wetland	0/1	*0%*	0/1	*0%*
Reference Pond	0/1	*0%*	0/1	*0%*

Non-detects were evaluated at one-half the reported detection limit.

Figure 5-3
Evaluation of Risks to Aquatic Receptors from Direct Contact with Dissolved Zinc in Surface Water

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site





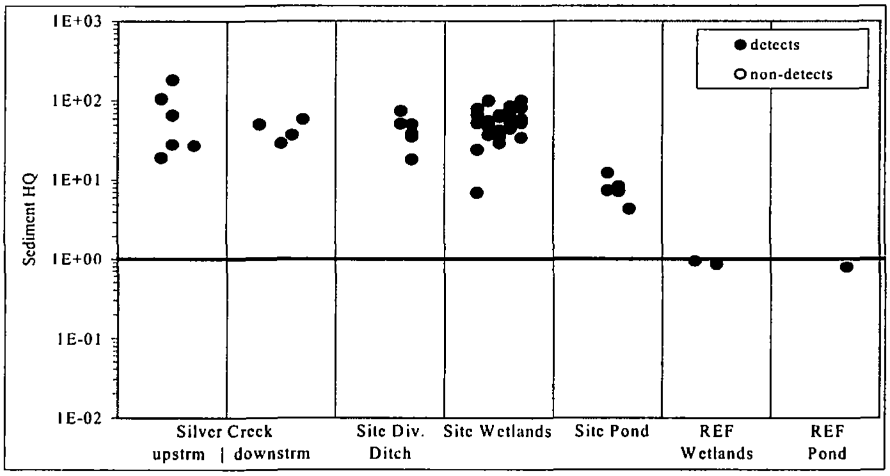
Location	N samples > benchmark			
	acute		chronic	
Silver Creek - upstream	84/110	76%	83/110	75%
Silver Creek - downstream	36/64	56%	36/64	56%
Site Diversion Ditch	16/59	27%	16/59	27%
Site Diversion Ditch - Wetlands Area	2/6	33%	2/6	33%
Site Pond	0/4	0%	0/4	0%
Unnamed Drainages	1/6	17%	1/6	17%
Reference Wetland	0/1	0%	0/1	0%
Reference Pond	0/1	0%	0/1	0%

Non-detects were evaluated at one-half the reported detection limit.

Figure 5-4
Evaluation of Risks to Benthic Invertebrates from Direct Contact with Cadmium and Copper in Bulk Sediment

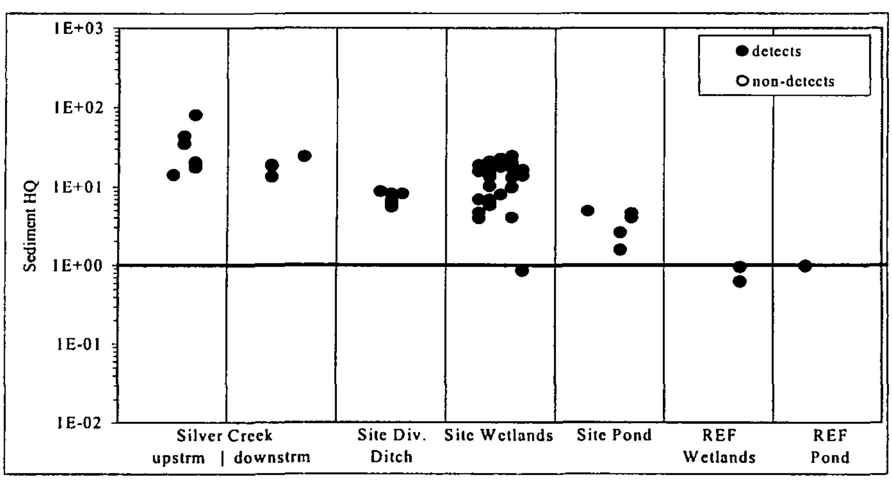
Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

CADMIUM



Location	% of samples > TEC benchmark	
Silver Creek - upstream	6/6	*100%*
Silver Creek - downstream	4/4	*100%*
Site Diversion Ditch	6/6	*100%*
Site Diversion Ditch - Wetlands Area	29/29	*100%*
Site Pond	5/5	*100%*
Reference Wetland	0/2	*0%*
Reference Pond	0/1	*0%*

COPPER

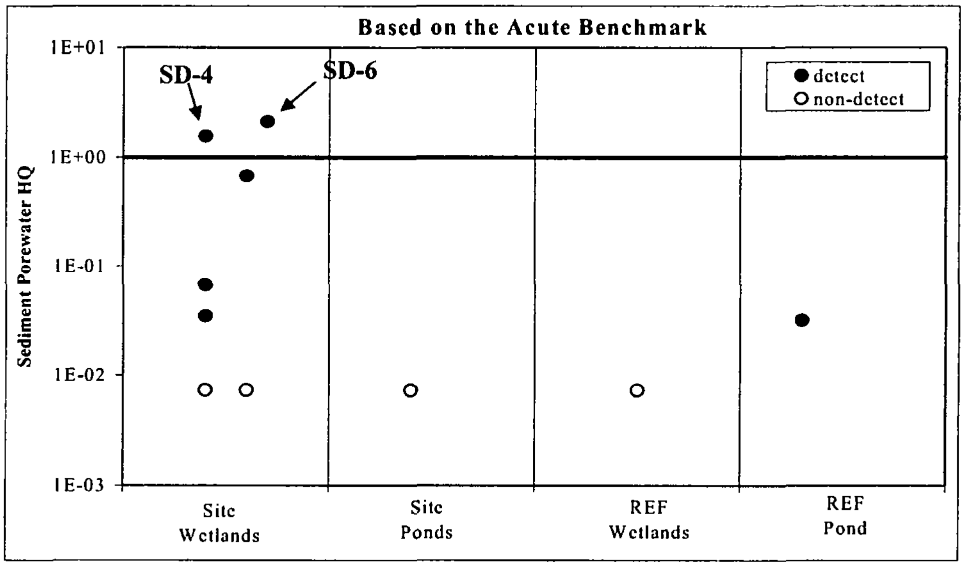


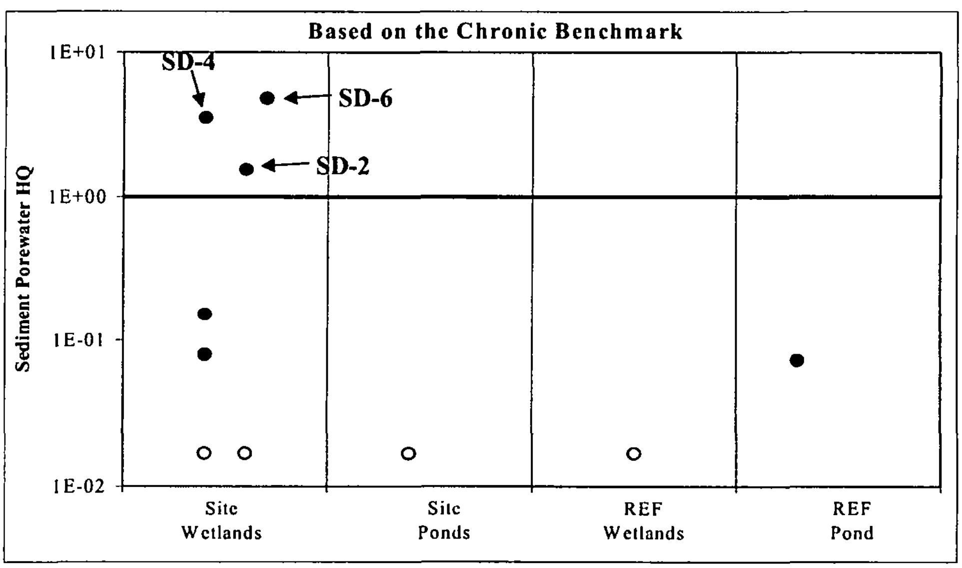
Location	% of samples > TEC benchmark	
Silver Creek - upstream	6/6	*100%*
Silver Creek - downstream	4/4	*100%*
Site Diversion Ditch	6/6	*100%*
Site Diversion Ditch - Wetlands Area	28/29	*97%*
Site Pond	5/5	*100%*
Reference Wetland	0/2	*0%*
Reference Pond	0/1	*0%*

Non-detects were evaluated at one-half the reported detection limit.

Figure 5-5
Screening-Level Evaluation of Risks to Aquatic Receptors from Direct Contact with Dissolved Arsenic in Sediment Porewater

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site



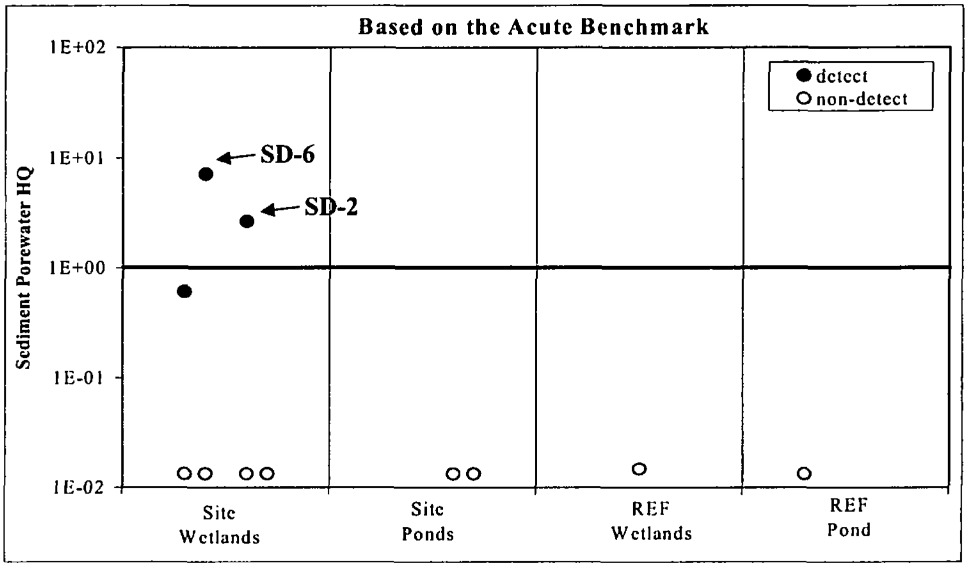


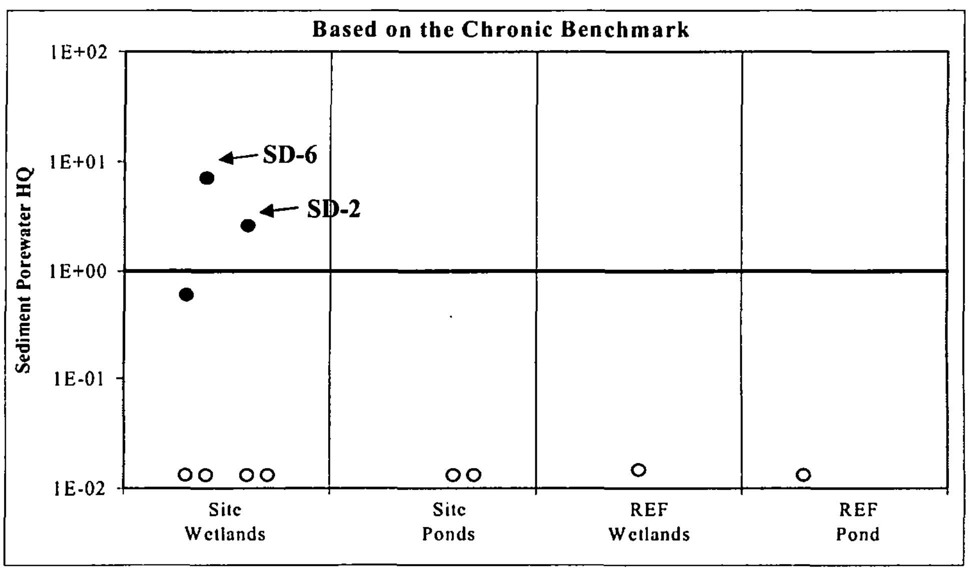
Location	N samples > benchmark			
	acute		chronic	
Site Diversion Ditch - Wetlands Area	2/8	25%	3/8	38%
Site Pond	0/2	0%	0/2	0%
Reference Wetland	0/1	0%	0/1	0%
Reference Pond	0/1	0%	0/1	0%

Non-detects were evaluated at one-half the reported detection limit.

Figure 5-6
Screening-Level Evaluation of Risks to Aquatic Receptors from Direct Contact with Dissolved Zinc in Sediment Porewater

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site



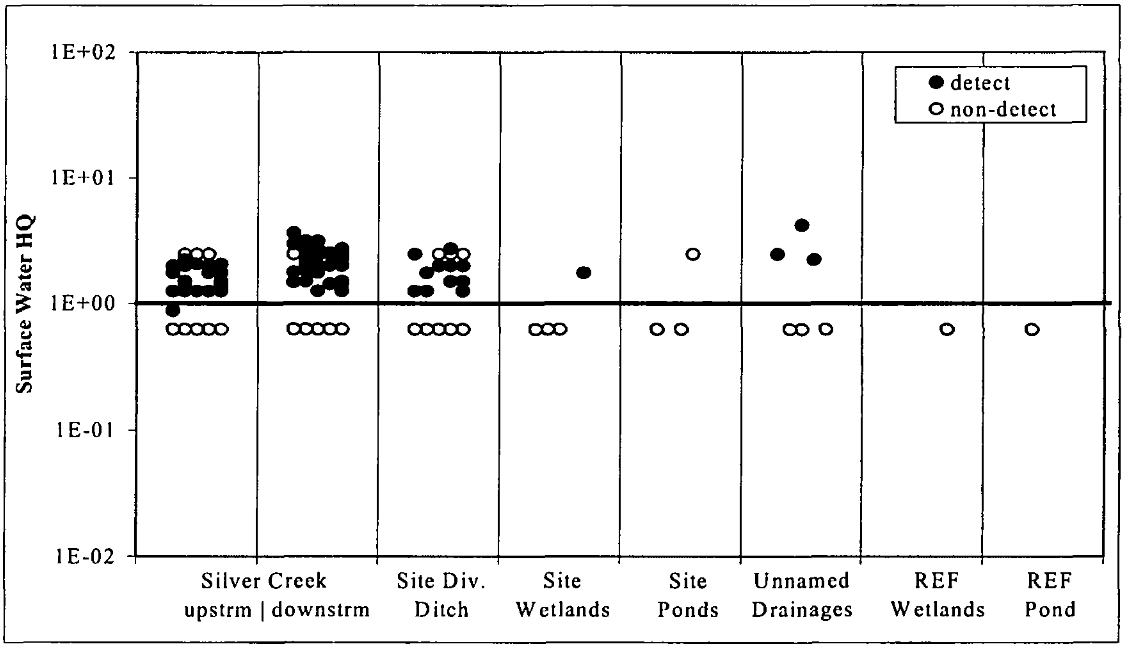


Location	N samples > benchmark			
	acute		chronic	
Site Diversion Ditch - Wetlands Area	2/8	*25%*	2/8	*25%*
Site Pond	0/2	*0%*	0/2	*0%*
Reference Wetland	0/1	*0%*	0/1	*0%*
Reference Pond	0/1	*0%*	0/1	*0%*

Non-detects were evaluated at one-half the reported detection limit.

Figure 6-1
Screening-Level Evaluation of Risks to Amphibians from Direct Contact with Dissolved Arsenic in Surface Water

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

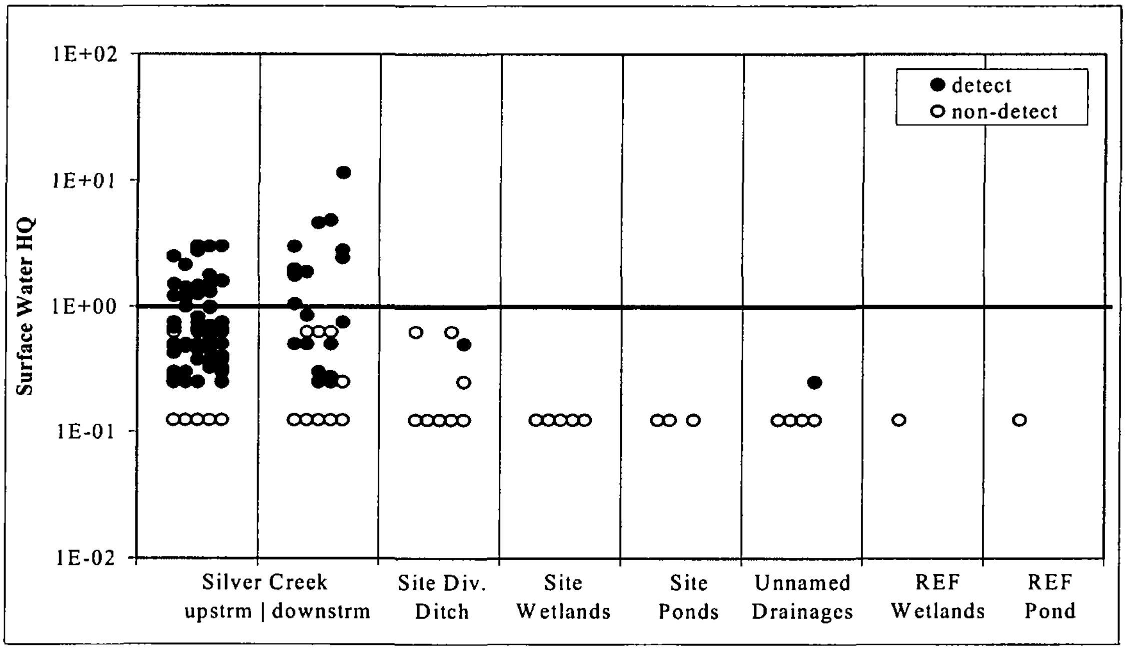


Location	N samples > benchmark	
Silver Creek - upstream	28/109	*26%*
Silver Creek - downstream	42/63	*67%*
Site Diversion Ditch	21/59	*36%*
Site Diversion Ditch - Wetlands Area	1/6	*17%*
Site Pond	0/4	*0%*
Unnamed Drainages	3/6	*50%*
Reference Wetland	0/1	*0%*
Reference Pond	0/1	*0%*

Non-detects were evaluated at one-half the reported detection limit.

Figure 6-2
Screening-Level Evaluation of Risks to Amphibians from Direct Contact with Dissolved Cadmium in Surface Water

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site



Location	N samples > benchmark	
Silver Creek - upstream	20/109	*18%*
Silver Creek - downstream	10/63	*16%*
Site Diversion Ditch	0/59	*0%*
Site Diversion Ditch - Wetlands Area	0/6	*0%*
Site Pond	0/4	*0%*
Unnamed Drainages	0/6	*0%*
Reference Wetland	0/1	*0%*
Reference Pond	0/1	*0%*

Non-detects were evaluated at one-half the reported detection limit.

Table 3-1
Summary of Samples Collected During the Phase I/II Field Investigations

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Station ID	Sediment	Surface Water	Sediment Porewater	Sediment Toxicity	Fish Tissue	Invert./ Snail Tissue	Plant Tissue
Site Wetland							
RFB-SD01	I					II, from two separate wetland reaches [a]	
RFB-SD02	I, II		II	II			II
RFB-SD03	I						
RFB-SD04	I, II	I	II	II			II
RFB-SD05	I						
RFB-SD06	I, II		II	II			II
RFB-SD07	I	I					
RFB-SD08	I						
RFB-SD09	I						
RFB-SD10	I, II		II	II			II
RFB-SD11	I, II		II	II			II
RFB-SD12	I						
RFB-SD13	I	I, II					
RFB-SD14	I, II		II	II			II
RFB-SD15	I, II		II	II			II
RFB-SD16	I						
RFB-SD17	I, II	I, II	II	II			II
Site Pond							
RFB-SD18	I, II	I, II	II	II	II, from entire pond	II, from entire pond	II
RFB-SD19	I						
RFB-SD20	I, II	I, II	II	II			II
Reference Wetland	I, II	II	II	II		II	II
Reference Pond	I, II	II	II	II		II	II

See Figure 3-1 for a map of site locations and Figure 3-3 for a map of reference locations.

I - Sampled as part of Phase I investigation in June 2003.
II - Sampled as part of Phase II investigation in August 2003.

[a] The upper wetland reach was located at station SD-6 near the Silver Creek inflow; the lower wetland reach was located along the south diversion ditch and included stations SD-13, SD-15, and SD-17.

Table 3-2
Exposure Area Descriptions for Aquatic and Terrestrial Habitats

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Exposure Area	**Description**
Aquatic Habitats	
Silver Creek - upstream & downstream of the RFT Site	A perennial stream which flows along the western site boundary; upstream/downstream designations are assigned at the rail trail bridge located northeast of State Highway 40 near the main embankment.
Site Diversion Ditches	*North Diversion Ditch* - collects snowmelt and storm water runoff from upslope, undisturbed areas north of the impoundment; flows in an easterly direction towards origin of the south diversion ditch. *South Diversion Ditch* - carries spring snowmelt and storm water runoff; flows from east to west and empties into Silver Creek just upstream of Highway 189 near the northern site boundary.
Site Wetlands Area	Wetlands located below the main embankment, near the confluence of the south diversion ditch with Silver Creek.
Site Pond	Small pond south of the wetlands area which receives water from the south diversion ditch.
Unnamed Drainages	Unnamed ephemeral drainages to the southeast of the main impoundment which flow into south diversion ditch.
Reference Wetland & Pond	Located in areas without mining activities, physical properties are similar to site wetland and pond habitats.
Terrestrial Habitats	
Tailings	Tailings from within the main impoundment and outside the impoundment.
On-Impoundment	Located on the main tailings impoundment; most areas have been covered with soil and revegetated as part of remediation activities.
Off-Impoundment	Located in areas north and south of the main impoundment potentially impacted by historically deposited and wind-blown tailings.
Background	Located in areas not expected to be affected by wind-blown RFT Site tailings; representative of anthropogenic levels (do not represent "pristine" levels).

Table 4-1 (Page 1 of 3)
Summary of Screening Level Ecological Risk Assessment Results

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Exposure Medium	Receptor	Exposure Pathway	Exposure Unit with Risks	COPCs	Range of HQ Values	Further Evaluation (Yes/No)
Surface Water	Aquatic Receptors	Direct Contact	Silver Creek upstream > Silver Creek downstream > South Diversion Ditch	Aluminum, arsenic, cadmium, chromium, copper, lead, mercury, selenium and zinc	HQ ≤1 to 200 (Total Acute) HQ ≤1 to 500 (Total Chronic) HQ ≤1 to 200 (Dissolved Acute) HQ ≤1 to 400 (Dissolved Chronic)	Yes for South Diversion Ditch and Wetlands
	Amphibians	Direct Contact	Silver Creek upstream > Silver Creek downstream > South Diversion Ditch > Unnamed drainage > ponded water Wetlands unknown	Arsenic, cadmium, copper, cyanide, lead, mercury and zinc	HQ ≤1 to 100,000	Yes for South Diversion Ditch and Wetlands
	Avian Wildlife	Ingestion	None	None	All HQs ≤1 (NOAEL) All HQs ≤1 (LOAEL)	No
	Mammalian Wildlife	Ingestion	Silver Creek Upstream	Lead	HQ ≤1 to 4 (NOAEL) All HQs ≤1 (LOAEL)	No
Seeps	Aquatic Receptors	Direct Contact	Groundwater at main embankment > upgradient groundwater	Aluminum, arsenic, cadmium, chromium, copper, cyanide, lead, mercury, selenium and zinc	HQ ≤1 to 500 (Total Acute) HQ ≤1 to 2,000 (Total Chronic) HQ ≤1 to 9 (Dissolved Acute) HQ ≤1 to 20 (Dissolved Chronic)	Yes
	Amphibians	Direct Contact	Groundwater at main embankment > upgradient groundwater	Arsenic, cadmium, copper, cyanide, lead, mercury, selenium, and zinc	HQ ≤1 to 50,000	Yes
	Plants	Direct Contact	Groundwater at main embankment > upgradient groundwater	Aluminum, arsenic, chromium, copper, lead, manganese, and zinc	HQ ≤1 to 300	Yes
	Avian Wildlife	Ingestion	None	None	All HQs ≤1 (NOAEL) All HQs ≤1 (LOAEL)	No
	Mammalian Wildlife	Ingestion	Upgradient groundwater	Lead	HQ ≤1 to 3 (NOAEL) All HQs ≤1 (LOAEL)	No

Table 4-1 (Page 2 of 3)
Summary of Screening Level Ecological Risk Assessment Results

Exposure Medium	Receptor	Exposure Pathway	Exposure Unit with Risks	COPCs	Range of HQ Values	Further Evaluation (Yes/No)
Sediment	Benthic Invertebrates	Direct Contact	Silver Creek upstream > Silver Creek downstream > South Diversion Ditch > Wetlands	Aluminum, antimony, arsenic, cadmium, chromium, copper, lead, manganese, mercury, nickel, silver, zinc	HQ ≤1 to 700 (Low Benchmark) HQ ≤1 to 300 (High Benchmark)	Yes for South Diversion Ditch and Wetlands
	Avian Wildlife	Incidental Ingestion	Silver Creek upstream >Silver Creek downstream > Wetlands area > South Diversion Ditch	Aluminum, arsenic, cadmium, lead, zinc	HQ ≤1 to 70 (NOAEL) HQ ≤1 to 30 (LOAEL)	Yes for Wetlands Area and South Diversion Ditch
	Mammalian Wildlife	Incidental Ingestion	Silver Creek Upstream >Silver Creek Downstream = Wetlands area > South Diversion Ditch	Aluminum, antimony, arsenic, lead, and thallium	HQ ≤1 to 60 (NOAEL) HQ ≤1 to 30 (LOAEL)	Yes for Wetlands Area and South Diversion Ditch
Soil	Plants	Direct Contact	Tailings > Off-impoundment > On-impoundment > background	Aluminum, antimony, arsenic, cadmium, chromium copper, lead, selenium, silver, zinc	HQ ≤1 to 500 (Low Benchmark) HQ ≤1 to 60 (High Benchmark)	Yes
	Soil Fauna	Direct Contact	Tailings > Off-impoundment > On-impoundment > background	Aluminum, arsenic, cadmium, chromium copper, lead, mercury, selenium, zinc	HQ ≤1 to 200 (Low Benchmark) HQ ≤1 to 5 (High Benchmark)	Yes
	Avian Wildlife	Incidental Ingestion	Tailings > On-impoundment > Off-impoundment > background	Aluminum, arsenic, barium, chromium, cadmium, copper, lead, mercury, selenium, and zinc	HQ ≤1 to 100 (NOAEL) HQ ≤1 to 50 (LOAEL)	Yes
	Mammalian Wildlife	Incidental Ingestion	Tailings > On-impoundment > Off-impoundment > background	Aluminum, antimony, arsenic, barium, cadmium, lead, selenium, and zinc	HQ ≤1 to 5,000 (NOAEL) HQ ≤1 to 2,000 (LOAEL)	Yes

Table 4-1 (Page 3 of 3)
Summary of Screening Level Ecological Risk Assessment Results

Exposure Medium	Receptor	Exposure Pathway	Exposure Unit with Risks	COPCs	Range of HQ Values	Further Evaluation (Yes/No)
Food Chain Items	Avian & Mammalian Piscivores	Ingestion of Fish	Silver Creek upstream > Silver Creek downstream > South Diversion Ditch > Wetlands	Aluminum, antimony, arsenic, barium, cadmium, chromium, cobalt, copper, lead, manganese, mercury, nickel, selenium, thallium, vanadium and zinc	HQ ≤1 to 20,000 (NOAEL) HQ ≤1 to 10,000 (LOAEL)	Yes for wetland and south diversion ditch
	Avian Aquatic Insectivores	Ingestion of Benthic Invertebrates	Silver Creek upstream > Silver Creek downstream > South Diversion Ditch > Wetlands	Aluminum, arsenic, barium, cadmium, chromium, cobalt, copper, lead, manganese, nickel, selenium and zinc	HQ ≤1 to 4,000 (NOAEL) HQ ≤1 to 600 (LOAEL)	Yes for wetland and south diversion ditch
	Avian & Mammalian Herbivores	Ingestion of Plants	Tailings > Off-impoundment soils > On-impoundment soils > Background	Antimony, lead, selenium, and zinc	HQ ≤1 to 30 (NOAEL) HQ ≤1 to 10 (LOAEL)	Yes
	Avian & Mammalian Terrestrial Insectivores	Ingestion of Terrestrial Invertebrates	Tailings > Off-impoundment soils > On-impoundment soils > Background	Arsenic, cadmium, copper, lead, mercury, selenium, and zinc	HQ ≤1 to 20,000 (NOAEL) HQ ≤1 to 5,000 (LOAEL)	Yes
	Avian & Mammalian Carnivores	Ingestion of Small Mammals	Tailings > Off-impoundment soils > On-impoundment soils > Background	Cadmium, chromium, lead, and selenium	HQ ≤1 to 200 (NOAEL) HQ ≤1 to 10 (LOAEL)	Yes

Table 4-2 (Page 1 of 3)
Summary of Data Gaps Identified in the Screening Level Ecological Risk Assessment

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Exposure Area	Data Type	Data Gaps	Data Collection
Wetland Area and Embankment	**Analytical Data**	Surface water data from the wetlands area were not available. Extent of contamination in surface water was unknown.	Collect surface water samples from wetland area and analyze for target analyte list (TAL) metals and water quality parameters.
		Sediment data from the wetland area were limited to four samples collected by E&E in 1993 (Table 3-9)	Collect additional sediment samples for analyses of TAL metals to better understand current extent of contamination after recent site activities. Complete concurrent analyses of metal concentrations in sediment porewater samples.
		Seep water data from the main embankment area were not available. Risks in the SLERA were estimated based on groundwater data. The location and extent of seeps along the embankment were not documented.	Collect seep samples and analyze for TAL metals. Locate and identify location and extent of seeps along the embankment.
	Biological Data	The type and extent of wetland habitat was not documented. Use of the wetland area by wildlife and aquatic receptors was unknown.	Collect qualitative information on the extent and nature of the wetlands habitat present including information on vegetative cover that would be used to identify possible use by wildlife and aquatic receptors. Complete a qualitative sampling of the wetlands area (concurrently with surface water, sediment and sediment porewater samples) to identify presence absence of macroinvertebrates and/or fish. Use by wildlife species should also be documented.

Table 4-2 (Page 2 of 3) Summary of Data Gaps Identified in the Screening Level Ecological Risk Assessment			
Exposure Area	**Data Type**	**Data Gaps**	**Data Collection**
Wetland Area and Embankment (cont.)	**Toxicological Data**	The SLERA predicted that surface water, seep water and sediments of the wetland area were toxic to aquatic receptors; however, site-specific toxicity was unknown.	Consider toxicity testing of seep water, sediment, and/or sediment porewater in consideration of habitat information obtained and site-specific needs to reduce the conservative screening estimates of the SLERA. Testing should be completed concurrently with sampling and analyses for analytical parameters and biological sampling.
	Biological Tissue Data	The SLERA predicted risks for wildlife species consuming, benthic invertebrates and fish from the wetlands area. The site-specific metals concentrations in food items was unknown.	Collect benthic organisms and fish (if present) from wetlands area for tissue analyses of TAL metals. Samples should be collected concurrently with other environmental media samples.
South Diversion Ditch	**Analytical Data**	Sampling of the sediments of the South Diversion Ditch was adequate for establishing extent of contamination. However, it may be necessary to collect further samples for analyses concurrently with any toxicity testing, benthic invertebrate sampling, or biological tissue sampling.	Collect concurrent analyes of TAL metals with any sediment, sediment porewater, benthic invertebrate community survey and/or biological tissue sampling. Complete sampling and analyses of TAL metals in sediment porewater to understand the bioavailability and potential toxicity of metals measured in bulk sediment samples.
	Biological Data	Specific information on the type of habitat provided by the South Diversion Ditch was not available. Potential use of the South Diversion Ditch by wildlife and aquatic receptors was unknown.	Collect qualitative information on the extent and nature of the habitat present including information on vegetative cover that would be used to identify possible use by wildlife and aquatic receptors. Complete a qualitative sampling of the wetlands area (concurrently with surface water, sediment and sediment pore water samples) to identify presence absence of macroinvertebrates and/or fish. Use by wildlife species should also be documented.

Table 4-2 (Page 3 of 3)
Summary of Data Gaps Identified in the Screening Level Ecological Risk Assessment

Exposure Area	Data Type	Data Gaps	Data Collection
South Diversion Ditch (cont.)	**Toxicological Data**	The SLERA predicted that surface water and sediments of the South Diversion ditch were toxic to aquatic receptors; however, site-specific toxicity is unknown.	Consider toxicity testing of sediment, and/or sediment pore water in consideration of habitat information obtained and site-specific needs to reduce the conservative screening estimates of the SLERA. Concurrent samples of media should be analyzed for TAL metals with analyses coordinated with any biological sampling or sampling of biological tissue.
	Biological Tissue Data	The SLERA predicted risks to wildlife species consuming benthic invertebrates and fish from the South Diversion Ditch. The site-specific metals concentrations in food items was unknown.	Collect benthic organisms and fish (if present) for tissue analyses of TAL metals.
On and Off Impoundment Soils	**Analytical Data**	Sampling of the soils on and off the main impoundment had analyzed for an inconsistent set of analytes.	Analyze future monitoring samples for TAL list. Analyze samples collected for concurrent analyses of tissues for TAL list.
	Biological Data	Specific information on the type of habitat provided by on-impoundment and off-impoundment areas was not available. Potential use of these areas by receptors was unknown.	Map and characterize the type of vegetative cover and soil cover off and on the main impoundment. Characterize habitat and identify possible terrestrial receptors (plants, invertebrates and wildlife).
	Toxicological Data	The SLERA predicted that on and off impoundment soils were potentially toxic to plants and soil invertebrates; however, site-specific toxicity was unknown.	Complete toxicity testing of soils with earthworms and/or plants to reduce the conservative screening estimates of the SERA. Testing should be completed concurrently with sampling and analyses for analytical parameters and biological sampling.
	Biological Tissue Data	The SLERA predicted risks to wildlife species consuming, plants, soil invertebrates and small mammals. The site-specific metals concentrations in food items was unknown.	Collect plants and soil invertebrates for tissue analyses of TAL metals.

Table 5-1
Surface Water Toxicity Benchmarks for Aquatic Receptors

Analyte Type	Analyte	ACUTE				CHRONIC				
		NAWQC - Acute (ug/L) [1]	GLWQI Tier II SAV (ug/L) [2]	USEPA R4 Acute (ug/L) [2]	**Surface Water Acute Benchmark (ug/L)**	NAWQC - Chronic (ug/L) [1]	GLWQI Tier II SCV (ug/L) [2]	USEPA R4 - Chronic (ug/L) [2]	Other (ug/L) [2]	**Surface Water Chronic Benchmark (ug/L)**
Inorganics	Aluminum	750 [6]	--	750	750	87	--	87	--	87
	Antimony	--	180	1300	180	--	30	160	--	30
	Arsenic	340 [9, 10]	--	360	340	150 [9, 10]	--	190	--	150
	Barium	50,000 [8]	110	--	50,000	5,000 [3]	--	--	--	5,000
	Beryllium	--	35	16	35	--	0.66	0.53	--	0.66
	Boron	--	30	--	30	--	1.6	750	7,000 EC20 Daphnids	1.60
	Cadmium	1.7 [4, 10]	--	3.92	1.72	0.22 [4, 10]	--	1.13	--	0.22
	Calcium	--	--	--	no benchmark	--	--	--	116,000 LCV Daphnids	116000
	Chromium III	499 [4, 10]	--	1,740	499	65 [4, 10]	--	207	--	65
	Chromium VI	16 [10]	--	16	16	10.6 [10]	--	11	--	11
	Cobalt	--	1,500	--	1,500	--	23	--	--	23
	Copper	12 [4, 10]	--	17.7	12	7.79 [4, 10]	--	11.8	--	8
	Cyanide	22 [12]	--	22	22	5.2 [12]	--	5.2	5	5.2
	Fluoride	--	--	--	no benchmark	--	--	--	1080 EC25 Bass Pop.	1080
	Fluorine	--	--	--	no benchmark	--	--	--	--	no benchmark
	Iron	--	--	--	no benchmark	1,000	--	1,000	300 CCME WQG	1,000
	Lead	54 [4, 10]	--	81.6	54	2.11 [4, 10]	--	3.18	--	2.1
	Lithium	--	260	--	260	--	14	--	--	14
	Magnesium	--	--	--	no benchmark	--	--	--	82,000 LCV Daphnids	82,000
	Manganese	--	2,300	--	2,300	--	120	--	--	120
	Mercury	1.2	--	2.4	1.2	0.65	1.3	0.012	--	0.65
	Molybdenum	--	16,000	--	16,000	--	370	--	--	370
	Nickel	408 [4, 10]	--	1420	408	45.3 [4, 10]	--	158	--	45
	Phosphorus	--	--	--	no benchmark	--	--	--	--	no benchmark
	Potassium	--	--	--	no benchmark	--	--	--	53,000 LCV Daphnids	53,000
	Selenium	--	--	20	20	5.0 [11]	--	5	--	5.0
	Silica	--	--	--	no benchmark	--	--	--	--	no benchmark
	Silver	3 [4, 10]	--	4.06	3	0.3 [3]	0.36	0.012	--	0.3
	Sodium	--	--	--	no benchmark	--	--	--	680,000 LCV Daphnids	680,000
	Strontium	--	15,000	--	15,000	--	1,500	--	--	1,500
	Sulfide	--	--	--	no benchmark	2.0	--	--	--	2.0
	Sulfur	--	--	--	no benchmark	--	--	--	--	no benchmark
	Thallium	--	110	140	110	--	12	4	--	12
	Vanadium	--	280	--	280	--	20	--	--	20
	Zinc	102 [4, 10]	--	117	102	102.94 [4, 10]	--	106	--	103

1 USEPA, 2002. National Recommended Water Quality Criteria: 2002. November 2002. EPA 822-R-02-047.
2 Spatial Analysis and Decision Assistance (SADA) Database version 3.0 - Table "Ecological SW Benchmarks"
3 Only acute NAWQC available; chronic NAWQC is equal to acute / 10.
4 Metal toxicity is hardness-dependent; values shown are calculated based on a hardness of 85 mg/L.
5 National Irrigation Water Quality Program (1998)
6 Aluminum NAWQC apply to waters with pH of 6.5 - 9.0.
7 Alkalinity NAWQC is the minimum required value.
8 Based on USEPA Gold Book value.
9 NAWQC derived from data for As 3+, but is applied here to total arsenic (this implies that As 3+ and As 5+ are equally toxic and their toxicities are additive).
10 NAWQC expressed in terms of the dissolved fraction.
11 NAWQC expressed in terms of the total recoverable fraction.
12 NAWQC expressed in terms of free cyanide.

NAWQC = National Ambient Water Quality Criteria
GLQWI = Great Lakes Water Quality Initiative
SAV/SCV = Secondary Acute/Chronic Value
CCME = Canadian Council of Ministers of the Environment
WQG = Water Quality Guidelines
LCV = Lowest Chronic Valuε
EC20 = Effect Concentration Causing Less Than 20% Reduction

Table 5-2
Selection of Surface Water COPCs for Aquatic Receptors

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Analyte	Detection Frequency		Mean Detection Limit (DL) (ug/L)	Max Detected Conc (ug/L)	Surface Water Benchmark (ug/L)	Is a Benchmark Available?	Is Analyte Detected?	Is Mean DL > Benchmark?	Is Max Detect > Benchmark?	COPC?
Aluminum, dissolved	25/206	*12%*	21	350	87	yes	yes	--	yes	YES
Antimony, dissolved	49/115	*43%*	2.5	15	30	yes	yes	--	no	NO
Arsenic, dissolved	83/262	*32%*	3.4	17	150	yes	yes	--	no	NO
Barium, dissolved	141/152	*93%*	50	520	5,000	yes	yes	--	no	NO
Beryllium, dissolved	0/12	*ND*	2.5	ND	0.66	yes	no	yes	--	Qual - Type 2
Boron, dissolved	5/13	*38%*	50	140	1.6	yes	yes	--	yes	YES
Cadmium, dissolved	106/259	*41%*	0.71	46	0.22	yes	yes	--	yes	YES
Calcium, dissolved	223/223	*100%*	NA	347,000	116,000	yes	yes	--	yes	YES
Chromium 6+, dissolved	1/13	*8%*	2.5	1.0	11	yes	yes	--	no	NO
Chromium, dissolved	18/254	*7%*	4.4	36	11	yes	yes	--	yes	YES
Cobalt, dissolved	0/12	*ND*	50	ND	23	yes	no	yes	--	Qual - Type 2
Copper, dissolved	28/256	*11%*	4.6	41	7.8	yes	yes	--	yes	YES
Cyanide, total	11/104	*11%*	2.4	54	5.2	yes	yes	--	yes	YES
Iron, dissolved	93/240	*39%*	36	1,000	1,000	yes	yes	--	no	NO
Lead, dissolved	36/265	*14%*	2.2	41	2.1	yes	yes	--	yes	YES
Magnesium, dissolved	223/223	*100%*	NA	184,000	82,000	yes	yes	--	yes	YES
Manganese, dissolved	233/237	*98%*	3	11,000	120	yes	yes	--	yes	YES
Mercury, dissolved	23/226	*10%*	6.0	0.22	0.65	yes	yes	--	no	NO
Phosphorus (P), dissolved	38/55	*69%*	0.010	3.4	no benchmark	no	--	--	--	Qual - Type 1
Potassium, dissolved	193/197	*98%*	750	33,100	53,000	yes	yes	--	no	NO
Selenium, dissolved	68/255	*27%*	1.9	6.0	5.0	yes	yes	--	yes	YES
Silica, dissolved	1/1	*100%*	NA	13	no benchmark	no	--	--	--	Qual - Type 1
Silver, dissolved	1/254	*0%*	2.2	5.0	0.26	yes	yes	--	yes	YES
Sodium, dissolved	197/197	*100%*	NA	5,330,000	680,000	yes	yes	--	yes	YES
Thallium, total	0/6	*ND*	0.80	ND	12	yes	no	no	--	NO
Vanadium, total	0/6	*ND*	18	ND	20	yes	no	no	--	NO
Zinc, dissolved	248/257	*96%*	8.9	83,000	103	yes	yes	--	yes	YES

NA = not applicable
ND = not detected

Table 5-3
Estimated Level of Concern for Aquatic Receptors from Direct Contact with Surface Water

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Exposure Area	Qualitative Level of Concern Based on the Distribution of Acute HQ Values (a)			
	Uncertain (b)	Low (c)	Moderate (d)	High (e)
Silver Creek - upstream	B, Mn	Al, Cd, Cr, Cu, CN, Pb, Se, Ag		Zn
Silver Creek - downstream	Mn	Al, Cd, Cr, Cu, CN, Pb, Se, Ag		Zn
Site Diversion Ditch	Mn	Al, Cd, Cr, Cu, CN, Pb, Se, Ag	Zn	
Site Diversion Ditch - Wetlands Area	B, Mn	Al, Cd, Cr, Cu, CN, Pb, Se, Ag, Zn		
Site Pond	B, Mn	Al, Cd, Cr, Cu, CN, Pb, Se, Ag, Zn		
Unnamed Drainages	Mn	Al, Cd, Cr, Cu, Pb, Se, Ag	Zn	
Reference Wetland	B, Mn	Al, Cd, Cr, Cu, CN, Pb, Se, Ag, Zn		
Reference Pond	B, Mn	Al, Cd, Cr, Cu, CN, Pb, Se, Ag, Zn		

Exposure Area	Qualitative Level of Concern Based on the Distribution of Chronic HQ Values			
	Uncertain	Low	Moderate	High
Silver Creek - upstream	B, Ca, Mn, Mg	Al, Cr, Cu, CN, Pb, Na, Se		Cd, Zn
Silver Creek - downstream	Ca, Mn, Mg	Al, Cd, Cr, Cu, CN, Pb, Na, Se		Cd, Zn
Site Diversion Ditch	Cd, Ca, Mn, Mg	Al, Cr, Cu, CN, Pb, Se	Zn	
Site Diversion Ditch - Wetlands Area	B, Ca, Mn, Mg	Al, Cd, Cr, Cu, CN, Pb, Se, Na, Zn		
Site Pond	B, Ca, Mn, Mg	Al, Cd, Cr, Cu, CN, Pb, Se, Na, Zn		
Unnamed Drainages	Cd, Mn	Al, Cr, Cu, Pb, Se	Zn	
Reference Wetland	B, Ca, Mn, Mg	Al, Cd, Cr, Cu, CN, Pb, Se, Na, Zn		
Reference Pond	B, Mn, Mg	Al, Cd, Ca, Cr, Cu, CN, Pb, Se, Na, Zn		

(a) The qualitative level of concern was assigned based on professional judgement, considering the exceedance frequency, the number of samples, the magnitude of the exceedance, and a comparison to reference, as discussed in Section 4.4.1.

(b) Risk is difficult to interpret either because of: inadequate detection limits (ie: HQs for non-detects exceed 1); overly conservative benchmarks (ie: HQs for reference areas exceed 1); or low number of samples, most of which have HQs close to 1.

(c) Risk is judged to be low if none or only a small fraction of HQs exceed 1 and the magnitude of the exceedances are small.

(d) Risk is judged to be moderate if a moderate fraction of HQs exceed 1 and the magnitude of the exceedances are mainly low.

(e) Risk is judged to be high if a large fraction of HQs exceed 1 and the magnitude of the exceedances are relatively high.

Table 5-4
Sediment Toxicity Benchmarks for Benthic Invertebrates

Analyte	Threshold Effect Concentrations (TEC)[1]				
	Consensus-Based TEC (mg/kg) [a]	ARCS TEL (mg/kg) [b]	Other (mg/kg)		Sediment Screening Benchmark (mg/kg)
Aluminum	--	25,519	--		25,519
Antimony	--	--	2.0	NOAA ERL [c]	2.0
Arsenic	9.8	11	--		9.8
Barium	--	--	--		no benchmark
Beryllium	--	--	--		no benchmark
Cadmium	0.99	0.58	--		1.0
Calcium	--	--	--		no benchmark
Chromium	43	36	--		43
Cobalt	--	--	--		no benchmark
Copper	32	28	--		32
Cyanide	--	--	--		no benchmark
Iron	--	188,400	--		188,400
Lead	36	37	--		36
Magnesium	--	--	--		no benchmark
Manganese	--	631	--		631
Mercury	0.18	--	--		0.18
Nickel	23	20	--		23
Potassium	--	--	--		no benchmark
Phosphorus	--	--	--		no benchmark
Selenium	--	--	--		no benchmark
Silver	--	--	1.0	NOAA ERL [c]	1
Sodium	--	--	--		no benchmark
Sulfide	--	--	--		no benchmark
Thallium	--	--	--		no benchmark
Vanadium	--	--	--		no benchmark
Zinc	121	98	--		121

Notes:

1 The TEC encompasses several types of sediment quality guidelines including the Lowest Effect Level (LEL), the Threshold Effect Level (TEL), the Effect Range Low (ERL), and the Minimum Effect Threshold (MET).

Sources Hierarchy:

a MacDonald et al. (2000); consensus-based threshold effect concentration (TEC).
b Ingersoll, et al. (1996); Threshold Effect Level (TEL) for total extraction of sediment (BT) samples from *Hyalella azteca* 28-day (HA28) tests.
c Long and Morgan (1990); NOAA Effect Range Low (ERL).

Table 5-5
Selection of Sediment COPCs for Benthic Invertebrates

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Analyte	Detection Frequency		Mean Detection Limit (DL) (mg/kg)	Max Detected Conc (mg/kg)	Sediment Benchmark (mg/kg)	Is a Benchmark Available?	Is Analyte Detected?	Is Mean DL > Benchmark?	Is Max Detect > Benchmark?	COPC?
Aluminum	53/53	*100%*	NA	28,800	25,519	yes	yes	--	yes	YES
Antimony	49/53	*92%*	5.0	889	2.0	yes	yes	--	yes	YES
Arsenic	53/53	*100%*	NA	1,735	9.8	yes	yes	--	yes	YES
Barium	37/37	*100%*	NA	2,562	no benchmark	no	--	--	--	Qual - Type 1
Beryllium	24/37	*65%*	0.5	2.3	no benchmark	no	--	--	--	Qual - Type 1
Cadmium	53/53	*100%*	NA	179	1.0	yes	yes	--	yes	YES
Calcium	4/4	*100%*	NA	96,000	no benchmark	no	--	--	--	Qual - Type 1
Chromium	53/53	*100%*	NA	68	43	yes	yes	--	yes	YES
Cobalt	37/37	*100%*	NA	68	no benchmark	no	--	--	--	Qual - Type 1
Copper	53/53	*100%*	NA	2,559	32	yes	yes	--	yes	YES
Iron	53/53	*100%*	NA	156,800	188,400	yes	yes	--	no	NO
Lead	53/53	*100%*	NA	42,990	36	yes	yes	--	yes	YES
Magnesium	4/4	*100%*	NA	14,100	no benchmark	no	--	--	--	Qual - Type 1
Manganese	37/37	*100%*	NA	161,000	631	yes	yes	--	yes	YES
Mercury	49/53	*92%*	0.020	6.2	0.18	yes	yes	--	yes	YES
Nickel	37/37	*100%*	NA	97	23	yes	yes	--	yes	YES
Phosphorus	13/13	*100%*	NA	5,363	no benchmark	no	--	--	--	Qual - Type 1
Potassium	4/4	*100%*	NA	4,760	no benchmark	no	--	--	--	Qual - Type 1
Selenium	40/53	*75%*	3.8	50	no benchmark	no	--	--	--	Qual - Type 1
Silver	52/53	*98%*	0.5	136	1.0	yes	yes	--	yes	YES
Sodium	4/4	*100%*	NA	1,150	no benchmark	no	--	--	--	Qual - Type 1
Sulfide	33/33	*100%*	NA	3,925	no benchmark	no	--	--	--	Qual - Type 1
Thallium	36/37	*97%*	1.3	50	no benchmark	no	--	--	--	Qual - Type 1
Vanadium	37/37	*100%*	NA	65	no benchmark	no	--	--	--	Qual - Type 1
Zinc	53/53	*100%*	NA	44,560	121	yes	yes	--	yes	YES

NA = not applicable
ND = not detected

COPC_Aq_Sediment.xls: ecosed_aquatic
1/30/2004

Table 5-6
Estimated Level of Concern for Aquatic Receptors from Direct Contact with Bulk Sediment

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Exposure Area	Qualitative Level of Concern Based on the Distribution of TEC HQ Values			
	Uncertain	Low	Moderate	High
Silver Creek - upstream	Sb, As, Pb, Ag, Zn	Al, Cr		Cd, Cu, Hg
Silver Creek - downstream	Sb, As, Pb, Ag, Zn	Al, Cr		Cd, Cu, Hg
Site Diversion Ditch	Sb, As, Pb, Ag, Zn	Al, Cr		Cd, Cu, Hg
Site Diversion Ditch - Wetlands Area	Sb, As, Pb, Mn, Ag, Zn	Al, Cr	Ni	Cd, Cu, Hg
Site Pond	Sb, As, Pb, Mn	Al, Cr, Ni		Cd, Cu, Hg, Ag, Zn

(a) The qualitative level of concern was assigned based on professional judgement, considering the exceedance frequency, the number of samples, the magnitude of the exceedance, and a comparison to reference, as discussed in Section 4.4.1.

(b) Risk is difficult to interpret either because of: inadequate detection limits (ie: HQs for non-detects exceed 1); overly conservative benchmarks (ie: HQs for reference areas exceed 1); or low number of samples, most of which have HQs close to 1.

(c) Risk is judged to be low if none or only a small fraction of HQs exceed 1 and the magnitude of the exceedances are small.

(d) Risk is judged to be moderate if a moderate fraction of HQs exceed 1 and the magnitude of the exceedances are mainly low.

(e) Risk is judged to be high if a large fraction of HQs exceed 1 and the magnitude of the exceedances are relatively high.

Table 5-7
Selection of Sediment Porewater COPCs for Benthic Invertebrates

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Analyte	Detection Frequency		Mean Detection Limit (DL) (ug/L)	Max Detected Conc (ug/L)	Screening-Level Porewater Benchmark (ug/L)	Is a Benchmark Available?	Is Analyte Detected?	Is Mean DL > Benchmark?	Is Max Detect > Benchmark?	COPC?
Aluminum, dissolved	0/12	*ND*	25	ND	87	yes	no	no	--	NO
Antimony, dissolved	5/14	*36%*	2.5	80	30	yes	yes	--	yes	YES
Arsenic, dissolved	6/12	*50%*	2.5	720	150	yes	yes	--	yes	YES
Barium, dissolved	5/12	*42%*	50	850	5,000	yes	yes	--	no	NO
Beryllium, dissolved	0/12	*ND*	2.5	ND	0.66	yes	no	yes	--	Qual - Type 2
Boron, dissolved	6/12	*50%*	50	240	1.6	yes	yes	--	yes	YES
Cadmium, dissolved	1/12	*8%*	0.50	5.0	0.59	yes	yes	--	yes	YES
Calcium, dissolved	12/12	*100%*	NA	458,000	116,000	yes	yes	--	yes	YES
Chromium 6+, dissolved	0/12	*ND*	2.5	ND	11	yes	no	no	--	NO
Chromium, dissolved	0/12	*ND*	5.0	ND	11	yes	no	no	--	NO
Cobalt, dissolved	0/12	*ND*	50	ND	23	yes	no	yes	--	Qual - Type 2
Copper, dissolved	1/12	*8%*	2.5	5.0	26.2	yes	yes	--	no	NO
Cyanide, total	0/12	*ND*	2.2	ND	5.2	yes	no	no	--	NO
Iron, dissolved	9/12	*75%*	50	17,000	1,000	yes	yes	--	yes	YES
Lead, dissolved	3/14	*21%*	2.5	110	9.6	yes	yes	--	yes	YES
Magnesium, dissolved	12/12	*100%*	NA	113,000	82,000	yes	yes	--	yes	YES
Manganese, dissolved	12/12	*100%*	NA	24,000	120	yes	yes	--	yes	YES
Mercury, dissolved	0/12	*ND*	100	ND	0.65	yes	no	yes	--	Qual - Type 2
Potassium, dissolved	8/12	*67%*	1,000	8,000	53,000	yes	yes	--	no	NO
Selenium, dissolved	0/12	*ND*	2.0	ND	5.0	yes	no	no	--	NO
Silver, dissolved	0/14	*ND*	2.3	ND	3.0	yes	no	no	--	NO
Sodium, dissolved	12/12	*100%*	NA	170,000	680,000	yes	yes	--	no	NO
Zinc, dissolved	3/12	*25%*	5.0	2,700	342	yes	yes	--	yes	YES

NA = not applicable
ND = not detected

Table 5-8
Estimated Level of Concern for Aquatic Receptors from Direct Contact with Sediment Porewater

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Exposure Area	Qualitative Level of Concern Based on the Distribution of Acute HQ Values (a)			
	Uncertain (b)	Low (c)	Moderate (d)	High (e)
Site Diversion Ditch - Wetlands Area	B, Mn	Sb, Cd, Cu, Pb	As	Zn
Site Pond	B, Mn	Sb, As, Cd, Cu, Pb, Zn		
Reference Wetland	B, Mn	Sb, As, Cd, Cu, Pb, Zn		
Reference Pond	B, Mn	Sb, As, Cd, Cu, Pb, Zn		

Exposure Area	Qualitative Level of Concern Based on the Distribution of Chronic HQ Values			
	Uncertain	Low	Moderate	High
Site Diversion Ditch - Wetlands Area	B, Ca, Fe, Mn	Cr, Cu, Ag, Mg	Sb, Cd, Pb	As, Zn
Site Pond	B, Ca, Mn	Sb, As, Cd, Cr, Cu, Fe, Pb, Mg, Ag, Zn		
Reference Wetland	B, Ca, Fe, Mn	Sb, As, Cd, Cr, Cu, Pb, Mg, Ag, Zn		
Reference Pond	B, Ca, Fe, Mn	Sb, As, Cd, Cr, Cu, Pb, Mg, Ag, Zn		

(a) The qualitative level of concern was assigned based on professional judgement, considering the exceedance frequency, the number of samples, the magnitude of the exceedance, and a comparison to reference, as discussed in Section 4.4.1.

(b) Risk is difficult to interpret either because of: inadequate detection limits (ie: HQs for non-detects exceed 1); overly conservative benchmarks (ie: HQs for reference areas exceed 1); or low number of samples, most of which have HQs close to 1.

(c) Risk is judged to be low if none or only a small fraction of HQs exceed 1 and the magnitude of the exceedances are small.

(d) Risk is judged to be moderate if a moderate fraction of HQs exceed 1 and the magnitude of the exceedances are mainly low.

(e) Risk is judged to be high if a large fraction of HQs exceed 1 and the magnitude of the exceedances are relatively high.

Table 5-9
Sediment Toxicity Results for the *Hyalella azteca* 28-day Test

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Sample ID	Survival (%)			Weight per Organism (mg dw)		
	Avg	Stdev		Avg	Stdev	
Lab Control	80	22		0.51	0.1	
Site Wetland						
RFB-TOX-SD2	0	0	† ‡	NA		
RFB-TOX-SD4	0	0	† ‡	NA		
RFB-TOX-SD6	0	0	† ‡	NA		
RFB-TOX-SD10	88	10		0.35	0.07	†
RFB-TOX-SD11	84	16		0.38	0.16	†
RFB-TOX-SD14	93	9		0.35	0.05	†
RFB-TOX-SD15	68	20	‡	0.19	0.07	† ‡
RFB-TOX-SD17	28	22	† ‡	0.06	0.03	† ‡
Site Pond						
RFB-TOX-SD18	96	5		0.57	0.11	
RFB-TOX-SD20	99	4		0.58	0.12	
Reference						
Reference pond	88	10		0.3	0.08	†
Reference wetland [a]	60	11	† ‡	0.26	0.11	†

† Statistically different compared to the lab control data.

‡ Statistically different compared to the reference pond sample data.

[a] *Data from the reference wetland sample were not used for statistical comparisons as the results did not meet the control performance criteria (at least 80% survival at termination).*

Table 5-10
Comparison of Tissue Burdens in Fish, Benthic Invertebrates, and Snails to Reference Concentrations and Adverse Effect Levels
Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Media	Fish Tissues				Increased Exposure?	Risk of Effects?
Location	Site Pond		Ref Pond	Effect Level Range		
Aluminum	75	44	na	8 - 36	–	yes
Antimony	0.17	0.11	na	9	–	no
Arsenic	0.53	<0.50	na	2.24 - 116	–	no
Barium	4.4	4.2	na	na	–	–
Cadmium	<0.50	<0.50	na	0.12 - 9.7	–	?
Chromium	<0.50	<0.50	na	na	–	–
Cobalt	<2.5	<2.5	na	na	–	–
Copper	1.7	1.3	na	11.1 - 42	–	no
Iron	151	105	na	na	–	–
Lead	7.9	4.6	na	0.4 - 4.0	–	yes
Manganese	165	173	na	na	–	–
Mercury	<0.020	<0.020	na	0.04 - 96.8	–	no
Nickel	<2.5	<2.5	na	na	–	–
Selenium	<1.0	<1.0	na	0.66 - 17.8	–	?
Silver	<0.10	<0.10	na	> 0.06	–	?
Thallium	<2.5	<2.5	na	na	–	–
Vanadium	<2.5	<2.5	na	2.22 - 3.12	–	?
Zinc	127	93	na	40 - 60	–	yes

Media	Snail Tissues				Increased Exposure?	Risk of Effects?
Location	Site Pond	Site Wetland, upper (b)	Ref Pond + Wetland	Effect Level Range		
Aluminum	21	122	54	na	yes	–
Antimony	0.35	1.1	<0.050	na	yes	–
Arsenic	0.72	3.1	0.68	na	yes	–
Barium	16	28	38	na	no	–
Cadmium	<0.50	<0.50	<0.50	30 - 125	no	no
Chromium	<0.50	<0.50	<0.50	na	no	–
Cobalt	<2.5	<2.5	<2.5	na	no	–
Copper	1.5	4.5	1.9	29.2 - 779	yes	no
Iron	122	782	677	na	yes	–
Lead	4.8	28	0.18	> 200	yes	no
Manganese	1563	1741	247	na	yes	–
Mercury	<0.020	<0.020	<0.020	3.28 - 4.66	no	no
Nickel	<2.5	<2.5	<2.5	na	no	–
Selenium	<1.0	<1.0	<1.0	0.22 - 29.6	no	?
Silver	<1.0	<1.0	<1.0	na	no	–
Thallium	<2.5	<2.5	<2.5	na	no	–
Vanadium	<2.5	<2.5	<2.5	na	no	–
Zinc	20	176	5.3	35.2 - 524	yes	yes

Media	Benthic Invertebrate Tissues						Increased Exposure?	Risk of Effects?
Location	Site Pond	Site Wetland, upper (b)	Site Wetland, lower (c)	Ref Pond	Ref Wetland	Effect Level Range		
Aluminum	<20	<20	<20	49	28	na	no	–
Antimony	<0.050	0.11	0.18	<0.050	<0.050	na	yes	–
Arsenic	<0.50	1.7	<0.50	<0.50	<0.50	na	yes	–
Barium	<2.5	<2.5	2.7	6	20	na	no	–
Cadmium	<0.50	<0.50	<0.50	<0.50	<0.50	3.5 - 134	no	no
Chromium	<0.50	<0.50	<0.50	<0.50	<0.50	na	no	–
Cobalt	<2.5	<2.5	<2.5	<2.5	<2.5	na	no	–
Copper	2.2	3.2	6	3.2	9.4	29.2 - 779	yes	no
Iron	29	108	99	202	337	na	no	–
Lead	1.4	4	4.5	0.11	0.16	98	yes	no
Manganese	23	10	141	23	238	na	no	–
Mercury	<0.020	<0.020	<0.020	<0.020	<0.020	3.28 - 4.66	no	no
Nickel	<2.5	<2.5	<2.5	<2.5	<2.5	na	no	–
Selenium	<1.0	<1.0	<1.0	<1.0	<1.0	0.22 - 29.6	no	?
Silver	<1.0	<1.0	<1.0	<1.0	<1.0	na	no	–
Thallium	<2.5	<2.5	<2.5	2.7	<2.5	na	no	–
Vanadium	<2.5	<2.5	<2.5	<2.5	<2.5	na	no	–
Zinc	26	49	44	19	17	35.2 - 524	yes	yes

All units are mg/kg ww.
Reported concentrations are based on the results from a single composite sample (N=1).
na = not available
? = inadequate detection limit, cannot determine risk

(a) See Appendix G for detailed information on reported Effect Level Ranges.
(b) Composite collected from reach near Silver Creek inflow at station SD-6.
(c) Composite collected from reach along south diversion ditch (includes stations SD-13, SD-15, SD-17).

Table 6-1
Screening-Level Toxicity Benchmarks for Amphibians from Aqueous Exposures

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Analyte	Species	Endpoint	Exposure Duration	Source	Lowest Value (ug/L)	Aqueous Screening Benchmark (ug/L)
Aluminum	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	7 days	Birge (1978)	50	5
Antimony	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	7 days	Birge (1978) & Birge et al. (1979)	300	30
Arsenic	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	7 days	Birge (1978) & Birge et al. (1979)	40	4.0
Beryllium	Spotted & Marbled Salamander (*Ambystoma* sp.)	LC50	2 - 4 days	Slonim and Ray (1975)	3150	315
Cadmium	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	7 days	Birge et al. (1979)	40	4.0
Chromium	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	7 days	Birge (1978) & Birge et al. (1979)	30	3.0
Cobalt	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	7 days	Birge (1978) & Birge et al. (1979)	50	5.0
Copper	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	7 days	Birge et al. (1979)	40	4.0
Lead	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	Not Reported	Birge et al. (1979)	40	4.0
Manganese	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	7 days	Birge (1978) & Birge et al. (1979)	1420	142
Mercury	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	7 days	Birge et al. (1979)	1	0.1
Nickel	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	7 days	Birge (1978) & Birge et al. (1979)	50	5.0
Selenium	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	7 days	Birge (1978) & Birge et al. (1979)	90	9.0
Silver	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	7 days	Birge (1978)	10	1.0
Zinc	Eastern Narrow-Mouthed Toad (*Gastrophryne carolinensis*)	LC50	7 days	Birge et al. (1979)	10	1.0

Lowest exposure concentration selected for screening benchmark.
Mercury benchmark is based on inorganic mercury.
For lethality endpoints, Screening Benchmark = LC50 / 10

Source: AQUIRE Database

Source Citations:

Birge, W.J. 1978. Aquatic Toxicology of Trace Elements of Coal and Fly Ash. In: J H Thorp and J W Gibbons (Eds.), Department of Energy Symposium Series, Energy and Environmental Stress in Aquatic Systems, Augusta, GA. 48:219-240.

Birge, W.J., J.E. Hudson, J.A. Black, and A.G. Westerman. 1979. Embryo-Larval Bioassays on Inorganic Coal Elements and in Situ Biomonitoring of Coal-Waste Effluents. In: Symposium US Fish & Wildlife Service, Surface Mining - Fish & Wildlife Needs in Eastern US, WV. 97-104.

Costa, H.H. 1965. Responses of Freshwater Animals to Sodium Cyanide Solutions III. Tadpoles of *Rana temporaria*. Ceylon J Sci Biol Sci 5(2):97-104.

Slonim, A.R. and E.E. Ray. 1975. Acute Toxicity of Beryllium Sulfate to Salamander Larvae (*Ambystoma* spp.). Bull Environ Contam Toxicol 13(3):307-312.

Table 6-2
Estimated Level of Concern for Amphibians from Direct Contact with Surface Water

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Exposure Area	Qualitative Level of Concern Based on the Distribution HQ Values (a)			
	Uncertain (b)	Low (c)	Moderate (d)	High (e)
Silver Creek - upstream	Al, Hg, Ag, Cr, Cu, Mn, Zn	Sb, CN, Se	Cd	As, Cu, Pb
Silver Creek - downstream	Al, Hg, Ag, Cr, Mn, Zn	Sb, CN, Cu, Se		As, Cd, Pb
Site Diversion Ditch	Al, Hg, Ag, Cr, Mn, Zn	Sb, Cd, CN, Pb, Se	Cu	As
Site Diversion Ditch - Wetlands Area	Al, Hg, Ag, Cr, Co, Mn, Zn	As, Sb, Be, Cd, CN, Cu, Pb, Se		
Site Pond	Al, As, Hg, Ag, Cr, Co, Mn, Zn	Sb, Be, Cd, CN, Cu, Pb, Se		
Unnamed Drainages	Al, Hg, Ag, Cr, Mn, Zn	Sb, Cd, Se, Pb	As, Cu	

(a) The qualitative level of concern was assigned based on professional judgement, considering the exceedance frequency, the number of samples, the magnitude of the exceedance, and a comparison to reference, as discussed in Section 4.4.1.

(b) Risk is difficult to interpret either because of: inadequate detection limits (ie: HQs for non-detects exceed 1); overly conservative benchmarks (ie: HQs for reference areas exceed 1); or low number of samples, most of which have HQs close to 1.

(c) Risk is judged to be low if none or only a small fraction of HQs exceed 1 and the magnitude of the exceedances are small.

(d) Risk is judged to be moderate if a moderate fraction of HQs exceed 1 and the magnitude of the exceedances are mainly low.

(e) Risk is judged to be high if a large fraction of HQs exceed 1 and the magnitude of the exceedances are relatively high.

Table 7-1
Exposure Factors for Representative Wildlife Species

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Receptor Class/Type		Surrogate Receptor	Body Weight (kg)	Food Ingestion Rate (kg wet weight/day)	Water Ingestion Rate (L/day)	Sediment Ingestion Rate (kg dry weight/day)	Home Range Size	Dietary Fraction (df)		
								Fish	Aquatic Invert.	Aquatic Plants
Bird	Omnivore	Mallard Duck	1.13	0.316	0.064	0.004	110 ha		0.75	0.25
Bird	Piscivore	Belted Kingfisher	0.147	0.073	0.016	0.0002	1.4 km (foraging distance)	1.00		
Bird	Insectivore	Cliff Swallow	0.023	0.013	0.005	0.00035	< 6 km (foraging radius)		1.00	
Mammal	Piscivore	Mink	0.556	0.089	0.058	0.0002	14 ha	1.00		

See Appendix I for detailed exposure factor and source information.

Table 7-2 (Page 1 of 2)
Exposure Point Concentrations (EPCs) Used to Evaluate Potential Risks to Wildlife

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Work Area	COPC	Exposure Point Concentrations (EPCs)				
		Surface Water	Sediment	Fish	Aquatic Invert.	Aquatic Plants
		mg/L	mg/kg dw	mg/kg ww	mg/kg ww	mg/kg ww
Site Diversion Ditch	Antimony	0.0034	89	0.17	1.1	0.9
	Arsenic	0.039	180	0.53	3.1	7.1
	Barium	0.15	na	4.4	28	24
	Beryllium	na	na	na	na	na
	Cadmium	0.0011	73	0.25	0.25	1
	Chromium	0.011	25	0.25	0.25	0.4
	Cobalt	na	na	1.2	1.2	1.4
	Copper	0.0061	270	1.7	6	5.9
	Lead	0.011	3100	7.9	28	21
	Manganese	5.8	na	170	1700	2400
	Mercury	0.00032	1.6	0.01	0.01	0.01
	Nickel	na	na	1.2	1.2	0.62
	Selenium	0.003	7.3	0.5	0.5	0.25
	Silver	0.0043	22	0.05	0.5	0.3
	Thallium	na	na	1.2	1.2	1.4
	Vanadium	na	na	1.2	1.2	0.62
	Zinc	0.69	12000	130	180	270
Site Diversion Ditch - Wetlands Area	Antimony	0.0025	110	0.17	1.1	0.85
	Arsenic	0.006	290	0.53	3.1	7.1
	Barium	0.05	410	4.4	28	15
	Beryllium	0.0025	0.79	na	na	na
	Cadmium	0.0023	69	0.25	0.25	1
	Chromium	0.005	42	0.25	0.25	0.32
	Cobalt	0.05	20	1.2	1.2	0.45
	Copper	0.007	610	1.7	6	4.4
	Lead	0.0057	5700	7.9	28	21
	Manganese	6.1	28000	170	1700	570
	Mercury	0.1	4.7	0.01	0.01	0.01
	Nickel	na	31	1.2	1.2	0.56
	Selenium	0.002	16	0.5	0.5	0.18
	Silver	0.0025	47	0.05	0.5	0.18
	Thallium	na	19	1.2	1.2	1.2
	Vanadium	na	24	1.2	1.2	0.45
	Zinc	1.1	12000	130	180	270
Site Pond	Antimony	0.0025	28	0.17	0.35	0.6
	Arsenic	0.004	60	0.53	0.72	1.4
	Barium	0.13	170	4.4	16	24
	Beryllium	0.0025	0.5	na	na	na
	Cadmium	0.0005	12	0.25	0.25	0.34
	Chromium	0.006	41	0.25	0.25	0.19
	Cobalt	0.05	22	1.2	1.2	1.4
	Copper	0.007	160	1.7	2.2	3.2
	Lead	0.0025	1500	7.9	4.8	4.2
	Manganese	1.9	5400	170	1600	2400
	Mercury	0.08	0.78	0.01	0.01	0.01
	Nickel	na	27	1.2	1.2	0.55
	Selenium	0.0021	4.9	0.5	0.5	0.14
	Silver	0.003	11	0.05	0.5	0.3
	Thallium	na	6.5	1.2	1.2	0.37
	Vanadium	na	26	1.2	1.2	0.31
	Zinc	0.022	3100	130	26	120

Table 7-2 (Page 2 of 2)
Exposure Point Concentrations (EPCs) Used to Evaluate Potential Risks to Wildlife

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Work Area	COPC	Exposure Point Concentrations (EPCs)				
		Surface Water	Sediment	Fish	Aquatic Invert.	Aquatic Plants
		mg/L	mg/kg dw	mg/kg ww	mg/kg ww	mg/kg ww
Reference Wetland	Antimony	0.0025	5	na	0.025	0.38
	Arsenic	0.0025	44	na	0.25	1.8
	Barium	0.59	2600	na	20	110
	Beryllium	0.0025	0.5	na	na	na
	Cadmium	0.0005	0.93	na	0.25	0.067
	Chromium	0.005	33	na	0.25	1.1
	Cobalt	0.05	68	na	1.2	1.8
	Copper	0.0025	30	na	9.4	3.1
	Lead	0.0025	82	na	0.16	3.7
	Manganese	5.7	70000	na	240	1500
	Mercury	0.1	0.01	na	0.01	0.01
	Nickel	na	22	na	1.2	1.5
	Selenium	0.002	43	na	0.5	0.13
	Silver	0.0025	75	na	0.5	0.13
	Thallium	na	41	na	1.2	0.34
	Vanadium	na	35	na	1.2	2.2
	Zinc	0.005	140	na	17	18
Reference Pond	Antimony	0.0025	5	na	0.025	na
	Arsenic	0.006	10	na	0.25	na
	Barium	0.38	460	na	6	na
	Beryllium	0.0025	0.5	na	na	na
	Cadmium	0.0005	0.78	na	0.25	na
	Chromium	0.005	30	na	0.25	na
	Cobalt	0.05	20	na	1.2	na
	Copper	0.0025	31	na	3.2	na
	Lead	0.0025	39	na	0.11	na
	Manganese	3.4	2100	na	23	na
	Mercury	0.1	0.01	na	0.01	na
	Nickel	na	20	na	1.2	na
	Selenium	0.002	5	na	0.5	na
	Silver	0.0025	0.5	na	0.5	na
	Thallium	na	1.2	na	2.7	na
	Vanadium	na	63	na	1.2	na
	Zinc	0.005	120	na	19	na

na = not available

Non-detects were evaluated at one-half the detection limit.

[ ] = measured tissue data are not available; EPC was assumed to be equal to the maximum measured concentration across all on-site locations.

Table 7-3
Summary of Selected Wildlife Toxicity Reference Values (TRVs)

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

COPC	Toxicity Reference Values (mg/kg BW/day)							
	Mammals				Birds			
	Low TRV/ NOAEL	High TRV/ LOAEL	**Estimated Threshold [a]**	Source	Low TRV/ NOAEL	High TRV/ LOAEL	**Estimated Threshold [a]**	Source
Aluminum			**narrative statement [b]**	1			**narrative statement [b]**	1
Antimony			**0.059**	1			**no TRV**	
Arsenic	0.32	4.7	**1.2**	2	5.5	22	**11**	2
Barium			**51.8**	1	21	42	**29**	3
Beryllium			**0.532**	1			**no TRV**	
Cadmium			**0.770**	1			**1.47**	1
Chromium	3.3	13.1	**6.6**	3 [c]	1.0	5.0	**2.2**	3 [c]
Cobalt			**7.34**	1			**7.61**	1
Copper	2.7	632	**41**	2	2.3	52	**11**	2
Lead			**4.70**	1			**1.63**	1
Manganese	14	159	**47**	2	78	776	**245**	2
Mercury, Inorganic	1.4	6.9	**3.1**	3	0.45	0.90	**0.64**	3
Mercury, Organic	0.25	4.0	**1.0**	2	0.039	0.180	**0.1**	2
Nickel	0.13	32	**2.1**	2	1.4	56	**8.8**	2
Selenium	0.05	1.21	**0.25**	2	0.23	0.93	**0.46**	2
Silver			**no TRV**				**no TRV**	
Thallium	0.48	1.43	**0.83**	2			**no TRV**	
Vanadium	0.21	2.1	**0.66**	3	11	--	**11**	3
Zinc	10	411	**63**	2	17	172	**54**	2

See Appendix C for details on the selected TRV.

[a] The estimated effects threshold is equal to the Eco-SSL TRV or is the geomean of the Low TRV/NOAEL and High TRV/LOAEL.

[b] Aluminum is expected to be a contaminant of potential concern only when pH is below 5.5.

[c] The mammalian TRV is based on Cr^{6+} (the lower of the Cr^{3+} and Cr^{6+} values). The bird TRV is based on Cr^{3+} (insufficient toxicity data in birds to derive a TRV for Cr^{6+}).

Source:
1 -- USEPA Eco-SSL (2003b)
2 -- Engineering Field Activity West (1998)
3 -- Sample et al. (1996)

Table 7-4 (Page 1 of 2)
Estimated Risks to the Mink from Ingestion of Contaminated Media

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Location	Analyte	Summary of Exposure Pathway HQs and Total HIs					
		Surface Water HQ	Sediment HQ	Fish HQ	Aquatic Invert. HQ	Aquatic Plants HQ	Total HI = ∑ HQ
Site Diversion Ditch	Antimony	<1	<1	<1	--	--	1
	Arsenic	<1	<1	<1	--	--	<1
	Barium	<1	--	<1	--	--	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	<1	--	--	<1
	Chromium	<1	<1	<1	--	--	<1
	Cobalt	--	--	<1	--	--	<1
	Copper	<1	<1	<1	--	--	<1
	Lead	<1	<1	<1	--	--	<1
	Manganese	<1	--	<1	--	--	<1
	Mercury	<1	<1	<1	--	--	<1
	Nickel	--	--	<1	--	--	<1
	Selenium	<1	<1	<1	--	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	<1	--	--	<1
	Vanadium	--	--	<1	--	--	<1
	Zinc	<1	<1	<1	--	--	<1
Site Diversion Ditch - Wetlands Area	Antimony	<1	<1	<1	--	--	1
	Arsenic	<1	<1	<1	--	--	<1
	Barium	<1	<1	<1	--	--	<1
	Beryllium	<1	<1	--	--	--	<1
	Cadmium	<1	<1	<1	--	--	<1
	Chromium	<1	<1	<1	--	--	<1
	Cobalt	<1	<1	<1	--	--	<1
	Copper	<1	<1	<1	--	--	<1
	Lead	<1	<1	<1	--	--	<1
	Manganese	<1	<1	<1	--	--	<1
	Mercury	<1	<1	<1	--	--	<1
	Nickel	--	<1	<1	--	--	<1
	Selenium	<1	<1	<1	--	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	<1	<1	--	--	<1
	Vanadium	--	<1	<1	--	--	<1
	Zinc	<1	<1	<1	--	--	<1
Site Pond	Antimony	<1	<1	<1	--	--	<1
	Arsenic	<1	<1	<1	--	--	<1
	Barium	<1	<1	<1	--	--	<1
	Beryllium	<1	<1	--	--	--	<1
	Cadmium	<1	<1	<1	--	--	<1
	Chromium	<1	<1	<1	--	--	<1
	Cobalt	<1	<1	<1	--	--	<1
	Copper	<1	<1	<1	--	--	<1
	Lead	<1	<1	<1	--	--	<1
	Manganese	<1	<1	<1	--	--	<1
	Mercury	<1	<1	<1	--	--	<1
	Nickel	--	<1	<1	--	--	<1
	Selenium	<1	<1	<1	--	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	<1	<1	--	--	<1
	Vanadium	--	<1	<1	--	--	<1
	Zinc	<1	<1	<1	--	--	<1

Table 7-4 (Page 2 of 2)
Estimated Risks to the Mink from Ingestion of Contaminated Media

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Location	Analyte	Summary of Exposure Pathway HQs and Total HIs					
		Surface Water HQ	Sediment HQ	Fish HQ	Aquatic Invert. HQ	Aquatic Plants HQ	Total HI = ∑ HQ
Reference Wetland	Antimony	<1	<1	--	--	--	<1
	Arsenic	<1	<1	--	--	--	<1
	Barium	<1	<1	--	--	--	<1
	Beryllium	<1	<1	--	--	--	<1
	Cadmium	<1	<1	--	--	--	<1
	Chromium	<1	<1	--	--	--	<1
	Cobalt	<1	<1	--	--	--	<1
	Copper	<1	<1	--	--	--	<1
	Lead	<1	<1	--	--	--	<1
	Manganese	<1	<1	--	--	--	<1
	Mercury	<1	<1	--	--	--	<1
	Nickel	--	<1	--	--	--	<1
	Selenium	<1	<1	--	--	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	<1	--	--	--	<1
	Vanadium	--	<1	--	--	--	<1
	Zinc	<1	<1	--	--	--	<1
Reference Pond	Antimony	<1	<1	--	--	--	<1
	Arsenic	<1	<1	--	--	--	<1
	Barium	<1	<1	--	--	--	<1
	Beryllium	<1	<1	--	--	--	<1
	Cadmium	<1	<1	--	--	--	<1
	Chromium	<1	<1	--	--	--	<1
	Cobalt	<1	<1	--	--	--	<1
	Copper	<1	<1	--	--	--	<1
	Lead	<1	<1	--	--	--	<1
	Manganese	<1	<1	--	--	--	<1
	Mercury	<1	<1	--	--	--	<1
	Nickel	--	<1	--	--	--	<1
	Selenium	<1	<1	--	--	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	<1	--	--	--	<1
	Vanadium	--	<1	--	--	--	<1
	Zinc	<1	<1	--	--	--	<1

-- = exposure pathway incomplete, or data (either toxicity or exposure data) are not available to calculate an HQ.
NC = Not Calculated

Table 7-5 (Page 1 of 2)
Estimated Risks to the Mallard Duck from Ingestion of Contaminated Media

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Location	Analyte	Summary of Exposure Pathway HQs and Total HIs					
		Surface Water HQ	Sediment HQ	Fish HQ	Aquatic Invert. HQ	Aquatic Plants HQ	Total HI = ∑ HQ
Site Diversion Ditch	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	--	<1	<1	<1
	Barium	<1	--	--	<1	<1	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	--	<1	<1	<1
	Chromium	<1	<1	--	<1	<1	<1
	Cobalt	--	--	--	<1	<1	<1
	Copper	<1	<1	--	<1	<1	<1
	Lead	<1	**6**	--	**4**	<1	**10**
	Manganese	<1	--	--	1	<1	**2**
	Mercury	<1	<1	--	<1	<1	<1
	Nickel	--	--	--	<1	<1	<1
	Selenium	<1	<1	--	<1	<1	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	--	--	<1	<1	<1
	Zinc	<1	<1	--	<1	<1	**2**
Site Diversion Ditch - Wetlands Area	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	--	<1	<1	<1
	Barium	<1	<1	--	<1	<1	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	--	<1	<1	<1
	Chromium	<1	<1	--	<1	<1	<1
	Cobalt	<1	<1	--	<1	<1	<1
	Copper	<1	<1	--	<1	<1	<1
	Lead	<1	**10**	--	**4**	<1	**20**
	Manganese	<1	<1	--	1	<1	**2**
	Mercury	<1	<1	--	<1	<1	<1
	Nickel	--	<1	--	<1	<1	<1
	Selenium	<1	<1	--	<1	<1	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	<1	--	<1	<1	<1
	Zinc	<1	<1	--	<1	<1	**2**
Site Pond	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	--	<1	<1	<1
	Barium	<1	<1	--	<1	<1	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	--	<1	<1	<1
	Chromium	<1	<1	--	<1	<1	<1
	Cobalt	<1	<1	--	<1	<1	<1
	Copper	<1	<1	--	<1	<1	<1
	Lead	<1	**3**	--	<1	<1	**4**
	Manganese	<1	<1	--	1	<1	**2**
	Mercury	<1	<1	--	<1	<1	<1
	Nickel	--	<1	--	<1	<1	<1
	Selenium	<1	<1	--	<1	<1	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	<1	--	<1	<1	<1
	Zinc	<1	<1	--	<1	<1	<1

Table 7-5 (Page 2 of 2)
Estimated Risks to the Mallard Duck from Ingestion of Contaminated Media

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Location	Analyte	Summary of Exposure Pathway HQs and Total HIs					
		Surface Water HQ	Sediment HQ	Fish HQ	Aquatic Invert. HQ	Aquatic Plants HQ	Total HI = ∑ HQ
Reference Wetland	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	--	<1	<1	<1
	Barium	<1	<1	--	<1	<1	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	--	<1	<1	<1
	Chromium	<1	<1	--	<1	<1	<1
	Cobalt	<1	<1	--	<1	<1	<1
	Copper	<1	<1	--	<1	<1	<1
	Lead	<1	<1	--	<1	<1	<1
	Manganese	<1	<1	--	<1	<1	**2**
	Mercury	<1	<1	--	<1	<1	<1
	Nickel	--	<1	--	<1	<1	<1
	Selenium	<1	<1	--	<1	<1	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	<1	--	<1	<1	<1
	Zinc	<1	<1	--	<1	<1	<1
Reference Pond	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	--	<1	--	<1
	Barium	<1	<1	--	<1	--	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	--	<1	--	<1
	Chromium	<1	<1	--	<1	--	<1
	Cobalt	<1	<1	--	<1	--	<1
	Copper	<1	<1	--	<1	--	<1
	Lead	<1	<1	--	<1	--	<1
	Manganese	<1	<1	--	<1	--	<1
	Mercury	<1	<1	--	<1	--	<1
	Nickel	--	<1	--	<1	--	<1
	Selenium	<1	<1	--	<1	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	<1	--	<1	--	<1
	Zinc	<1	<1	--	<1	--	<1

-- = exposure pathway incomplete, or data (either toxicity or exposure data) are not available to calculate an HQ.
NC = Not Calculated

Table 7-6 (Page 1 of 2)
Estimated Risks to the Belted Kingfisher from Ingestion of Contaminated Media

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Location	Analyte	Summary of Exposure Pathway HQs and Total HIs					
		Surface Water HQ	Sediment HQ	Fish HQ	Aquatic Invert. HQ	Aquatic Plants HQ	Total HI = ∑ HQ
Site Diversion Ditch	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	<1	--	--	<1
	Barium	<1	--	<1	--	--	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	<1	--	--	<1
	Chromium	<1	<1	<1	--	--	<1
	Cobalt	--	--	<1	--	--	<1
	Copper	<1	<1	<1	--	--	<1
	Lead	<1	3	2	--	--	5
	Manganese	<1	--	<1	--	--	<1
	Mercury	<1	<1	<1	--	--	<1
	Nickel	--	--	<1	--	--	<1
	Selenium	<1	<1	<1	--	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	--	<1	--	--	<1
	Zinc	<1	<1	1	--	--	1
Site Diversion Ditch - Wetlands Area	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	<1	--	--	<1
	Barium	<1	<1	<1	--	--	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	<1	--	--	<1
	Chromium	<1	<1	<1	--	--	<1
	Cobalt	<1	<1	<1	--	--	<1
	Copper	<1	<1	<1	--	--	<1
	Lead	<1	5	2	--	--	7
	Manganese	<1	<1	<1	--	--	<1
	Mercury	<1	<1	<1	--	--	<1
	Nickel	--	<1	<1	--	--	<1
	Selenium	<1	<1	<1	--	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	<1	<1	--	--	<1
	Zinc	<1	<1	1	--	--	1
Site Pond	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	<1	--	--	<1
	Barium	<1	<1	<1	--	--	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	<1	--	--	<1
	Chromium	<1	<1	<1	--	--	<1
	Cobalt	<1	<1	<1	--	--	<1
	Copper	<1	<1	<1	--	--	<1
	Lead	<1	1	2	--	--	4
	Manganese	<1	<1	<1	--	--	<1
	Mercury	<1	<1	<1	--	--	<1
	Nickel	--	<1	<1	--	--	<1
	Selenium	<1	<1	<1	--	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	<1	<1	--	--	<1
	Zinc	<1	<1	1	--	--	1

Table 7-6 (Page 2 of 2)
Estimated Risks to the Belted Kingfisher from Ingestion of Contaminated Media

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Location	Analyte	Summary of Exposure Pathway HQs and Total HIs					
		Surface Water HQ	Sediment HQ	Fish HQ	Aquatic Invert. HQ	Aquatic Plants HQ	Total HI = ∑ HQ
Reference Wetland	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	--	--	--	<1
	Barium	<1	<1	--	--	--	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	--	--	--	<1
	Chromium	<1	<1	--	--	--	<1
	Cobalt	<1	<1	--	--	--	<1
	Copper	<1	<1	--	--	--	<1
	Lead	<1	<1	--	--	--	<1
	Manganese	<1	<1	--	--	--	<1
	Mercury	<1	<1	--	--	--	<1
	Nickel	--	<1	--	--	--	<1
	Selenium	<1	<1	--	--	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	<1	--	--	--	<1
	Zinc	<1	<1	--	--	--	<1
Reference Pond	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	--	--	--	<1
	Barium	<1	<1	--	--	--	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	--	--	--	<1
	Chromium	<1	<1	--	--	--	<1
	Cobalt	<1	<1	--	--	--	<1
	Copper	<1	<1	--	--	--	<1
	Lead	<1	<1	--	--	--	<1
	Manganese	<1	<1	--	--	--	<1
	Mercury	<1	<1	--	--	--	<1
	Nickel	--	<1	--	--	--	<1
	Selenium	<1	<1	--	--	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	<1	--	--	--	<1
	Zinc	<1	<1	--	--	--	<1

-- = exposure pathway incomplete, or data (either toxicity or exposure data) are not available to calculate an HQ.
NC = Not Calculated

Table 7-7 (Page 1 of 2)
Estimated Risks to the Cliff Swallow from Ingestion of Contaminated Media

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Location	Analyte	Summary of Exposure Pathway HQs and Total HIs					
		Surface Water HQ	Sediment HQ	Fish HQ	Aquatic Invert. HQ	Aquatic Plants HQ	Total HI = ∑ HQ
Site Diversion Ditch	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	--	<1	--	<1
	Barium	<1	--	--	<1	--	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	--	<1	--	<1
	Chromium	<1	<1	--	<1	--	<1
	Cobalt	--	--	--	<1	--	<1
	Copper	<1	<1	--	<1	--	<1
	Lead	<1	**30**	--	**9**	--	**40**
	Manganese	<1	--	--	**4**	--	**4**
	Mercury	<1	<1	--	<1	--	<1
	Nickel	--	--	--	<1	--	<1
	Selenium	<1	<1	--	<1	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	--	--	<1	--	<1
	Zinc	<1	**3**	--	**2**	--	**5**
Site Diversion Ditch - Wetlands Area	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	--	<1	--	<1
	Barium	<1	<1	--	<1	--	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	--	<1	--	<1
	Chromium	<1	<1	--	<1	--	<1
	Cobalt	<1	<1	--	<1	--	<1
	Copper	<1	<1	--	<1	--	1
	Lead	<1	**50**	--	**9**	--	**60**
	Manganese	<1	**2**	--	**4**	--	**5**
	Mercury	<1	<1	--	<1	--	<1
	Nickel	--	<1	--	<1	--	<1
	Selenium	<1	<1	--	<1	--	1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	<1	--	<1	--	<1
	Zinc	<1	**3**	--	**2**	--	**5**
Site Pond	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	--	<1	--	<1
	Barium	<1	<1	--	<1	--	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	--	<1	--	<1
	Chromium	<1	<1	--	<1	--	<1
	Cobalt	<1	<1	--	<1	--	<1
	Copper	<1	<1	--	<1	--	<1
	Lead	<1	**10**	--	**2**	--	**20**
	Manganese	<1	<1	--	**4**	--	**4**
	Mercury	<1	<1	--	<1	--	<1
	Nickel	--	<1	--	<1	--	<1
	Selenium	<1	<1	--	<1	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	<1	--	<1	--	<1
	Zinc	<1	<1	--	<1	--	1

Table 7-7 (Page 2 of 2)
Estimated Risks to the Cliff Swallow from Ingestion of Contaminated Media

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Location	Analyte	Summary of Exposure Pathway HQs and Total HIs					
		Surface Water HQ	Sediment HQ	Fish HQ	Aquatic Invert. HQ	Aquatic Plants HQ	Total HI = ∑ HQ
Reference Wetland	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	--	<1	--	<1
	Barium	<1	1	--	<1	--	**2**
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	--	<1	--	<1
	Chromium	<1	<1	--	<1	--	<1
	Cobalt	<1	<1	--	<1	--	<1
	Copper	<1	<1	--	<1	--	<1
	Lead	<1	<1	--	<1	--	<1
	Manganese	<1	4	--	<1	--	**5**
	Mercury	<1	<1	--	<1	--	<1
	Nickel	--	<1	--	<1	--	<1
	Selenium	<1	1	--	<1	--	**2**
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	<1	--	<1	--	<1
	Zinc	<1	<1	--	<1	--	<1
Reference Pond	Antimony	--	--	--	--	--	NC
	Arsenic	<1	<1	--	<1	--	<1
	Barium	<1	<1	--	<1	--	<1
	Beryllium	--	--	--	--	--	NC
	Cadmium	<1	<1	--	<1	--	<1
	Chromium	<1	<1	--	<1	--	<1
	Cobalt	<1	<1	--	<1	--	<1
	Copper	<1	<1	--	<1	--	<1
	Lead	<1	<1	--	<1	--	<1
	Manganese	<1	<1	--	<1	--	<1
	Mercury	<1	<1	--	<1	--	<1
	Nickel	--	<1	--	<1	--	<1
	Selenium	<1	<1	--	<1	--	<1
	Silver	--	--	--	--	--	NC
	Thallium	--	--	--	--	--	NC
	Vanadium	--	<1	--	<1	--	<1
	Zinc	<1	<1	--	<1	--	<1

-- = exposure pathway incomplete, or data (either toxicity or exposure data) are not available to calculate an HQ.
NC = Not Calculated

Table 7-8
Primary Drivers of Predicted Risks in Wildlife Receptors

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Receptors	Primary Risk Drivers						
	Contaminants [a]	Exposure Areas [b]	Exposure Pathways (Range of HQs > 1)				
			Food		Sediment		Surface Water
Site Diversion Ditch, Wetlands Area, Site Pond							
Mink	*none*	*none*	*Total HIs for all contaminants ≤ 1*				
Belted Kingfisher	lead	wetlands > south diversion ditch > pond	○	(2 [c])	●	(3 - 5)	< 1
Mallard Duck	lead	wetlands > south diversion ditch > pond	○	(4)	●	(3 - 10)	< 1
	zinc	wetlands = south diversion ditch = pond	*All individual media HQs ≤ 1 (Total HI = 2)*				
	manganese	wetlands = south diversion ditch = pond	*All individual media HQs ≤ 1 (Total HI = 2)*				
Cliff Swallow	lead	wetlands > south diversion ditch > pond	○	(2 - 9)	●	(10 - 50)	< 1
	manganese	wetlands > pond	○	(2)	●	(4)	< 1
	zinc	wetlands > south diversion ditch	○	(2)	●	(3)	< 1

● = Primary contributor
○ = Secondary contributor

[a] Primary contaminants relative to reference locations.

[b] Shown in order of highest predicted risks to lowest predicted risks.

[c] Reference data are not available for fish tissue.

Table 8-1
Summary of Uncertainties in the Baseline Ecological Risk Assessment

Baseline Ecological Risk Assessment for the Richardson Flat Tailings Site

Assessment Component	Description	Likely Direction of Error	Likely Magnitude of Error
Nature and Extent of Contamination	Samples collected may not be fully representative of variability in space or time, especially if the number of samples is small.	Unknown	Probably small
	Analytical results may be imprecise.	Unknown	Probably small
Exposure Assessment	Some exposure pathways were not evaluated.	Underestimate of risk	Probably small
	Some chemicals were not evaluated because chemical was never detected, but detection limit was too high to detect the chemical if it were present at a level of concern.	Underestimate of risk	Usually small
	Exposure parameters for wildlife receptors are based on studies at other sites.	Unknown	Probably small
	Exposure point concentrations for wildlife receptors are based on a conservative estimate of the mean concentration in the exposure area.	Overestimate of risks	Possibly significant
	Absorption from site media is assumed to be the same as in laboratory studies.	Overestimate of risks	Possibly significant
Toxicity Assessment	Many chemicals lack reliable toxicity benchmarks for some receptors for some media; these chemicals are not evaluated.	Underestimation of risk	Probably small in most cases
	Available toxicity benchmarks are often based on limited data, and values must be extrapolated across species.	Unknown	Unknown, could be significant
	Wildlife receptors selected as representative species may not capture the full range of sensitivities in site receptors.	Unknown	Probably small
	Aquatic toxicity benchmarks are based on a wide range of species, some of which do not occur at this site.	Likely to overestimate risk	Probably small
Risk Characterization	Interactions between chemicals are difficult to account for; effects of one chemical may increase, decrease, or have no effect on other chemicals.	Unknown	Unknown, but probably small
	Estimation of population-level effects from HQ calculations is difficult and subject to professional judgement.	Unknown	Unknown, probably small in most cases